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**REMEDIAL INVESTIGATION/FEASIBILITY STUDY
CONRAIL SITE
ELKHART, INDIANA**

**REMEDIAL INVESTIGATION REPORT
VOLUME 1 OF 2**

**ARCS CONTRACT NO. 68-W8-0086
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Prepared for:

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TABLE OF CONTENTS VOLUME 1

<u>Section</u>		<u>Page</u>
	EXECUTIVE SUMMARY	1
1	INTRODUCTION AND SITE BACKGROUND	1-1
	1.1 SITE DESCRIPTION	1-2
	1.2 SITE PHYSICAL FEATURES	1-2
	1.3 LAND USE AND POPULATION	1-3
	1.4 GROUNDWATER RESOURCES, SURFACE WATER RESOURCES, AND CLIMATE	1-4
	1.5 SITE HISTORY AND PREVIOUS INVESTIGATIONS	1-5
2	PHASE III FIELD INVESTIGATION PROCEDURES	2-1
	2.1 LEAD-SCREEN AUGER SAMPLING	2-1
	2.2 SOIL SAMPLING	2-3
	2.3 MONITORING WELL INSTALLATION	2-4
	2.4 MONITORING WELL SURVEYING AND GROUNDWATER SAMPLING	2-6
	2.5 DRAINAGE NETWORK SAMPLING	2-8
	2.6 SURFACE WATER AND SEDIMENT SAMPLING	2-9
	2.7 INVESTIGATION-DERIVED WASTE	2-10
3	PHYSICAL CHARACTERISTICS OF THE SITE	3-1
	3.1 GEOLOGY	3-1
	3.2 HYDROGEOLOGY	3-4
4	NATURE AND EXTENT OF CONTAMINATION	4-1
	4.1 LEAD-SCREEN AUGER SAMPLING RESULTS	4-1
	4.1.1 County Road/Plume	4-2
	4.1.2 LaRue Street Plume	4-5
	4.2 SOIL SAMPLING RESULTS	4-7
	4.2.1 Track 69 Source Area	4-8
	4.2.2 Track 65 and 66 Source Area	4-8
	4.2.3 Other Potential Source Areas	4-9
	4.2.4 Discussion	4-9
	4.3 GROUNDWATER MONITORING WELL SAMPLING RESULTS	4-11
	4.3.1 County Road 1 Plume	4-12

Table of Contents (Cont.)

<u>Section</u>		<u>Page</u>
	4.3.2 Discussion of the County Road 1 Plume	4-14
	4.3.3 LaRue Street Plume	4-15
	4.3.4 Discussion of the LaRue Street Plume	4-16
4.4	DRAINAGE NETWORK SAMPLING RESULTS	4-17
4.5	SURFACE WATER AND SEDIMENT SAMPLING RESULTS	4-18
5	FATE AND TRANSPORT OF CHEMICALS OF POTENTIAL CONCERN	5-1
5.1	GENERAL FACTORS INFLUENCING TRANSPORT AND FATE AT THE SITE	5-1
	5.1.1 Volatilization	5-2
	5.1.2 Liquid Transport	5-3
	5.1.3 Sorption and Retardation	5-6
	5.1.4 Transformation Reactions	5-8
5.2	SITE-SPECIFIC FATE AND TRANSPORT MECHANISMS	5-9
	5.2.1 Movement, Mass, and Loading of Chemicals of Potential Concern	5-9
	5.2.1.1 Travel Time	5-9
	5.2.1.2 Contaminant Mass	5-10
	5.2.1.3 Chemical Loading to St. Joseph River	5-12
	5.2.1.4 Natural Attenuation	5-14
	5.2.2 Site-Specific Transformation Processes	5-15
6	RISK ASSESSMENT	6-1
6.1	CONCEPTUAL SITE MODEL	6-2
	6.1.1 Current Site Conditions	6-2
	6.1.2 Future Site Conditions	6-3
	6.1.3 Organization of the Risk Assessment	6-3
6.2	IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN	6-4
	6.2.1 Data Collection	6-4
	6.2.2 Data Evaluation	6-5
	6.2.2.1 Data Validation	6-5
	6.2.2.2 Quantitation Limits	6-6
	6.2.2.3 Data Qualifiers	6-6
	6.2.2.4 Blanks	6-7
	6.2.2.5 Background Concentrations	6-8
	6.2.2.6 Previous and Current Phase Data Reporting	6-8
	6.2.2.7 Selection of Chemicals of Potential Concern	6-8

y

Table of Contents (Cont.)

<u>Section</u>	<u>Page</u>
6.4.4.5 Summary of Toxicity Uncertainties	6-49
6.5 RISK CHARACTERIZATION	6-50
6.5.1 Introduction	6-50
6.5.2 Summary Discussion of the Risk Characterization . .	6-51
6.5.2.1 Characterization of Contamination	
Present at the Site	6-51
6.5.2.2 Risk Assessment Benchmarks	6-51
6.5.2.3 Magnitude and Sources of Risks Posed	
by Site Contaminants	6-52
6.5.2.4 Nature of Potential Adverse Health	
Effects	6-54
6.5.2.5 Level of Confidence/Uncertainty in the	
Risk Estimates	6-55
6.5.2.6 Major Factors Driving Risks	6-55
6.5.3 Risk Characterization Uncertainties	6-56
6.6 ECOLOGICAL ASSESSMENT	6-58
6.7 CHARACTERIZATION OF NATURAL FEATURES	
IN THE VICINITY OF THE SITE	6-60
6.7.1 Introduction	6-60
6.7.2 Characterization of Natural Habitats	6-60
6.7.2.1 Conrail Site	6-60
6.7.2.2 St. Joseph River	6-61
6.7.2.3 Baugo Creek and Baugo Bay	6-62
6.7.2.4 Ponds and Wetlands	6-63
6.8 CHEMICALS OF POTENTIAL ECOLOGICAL CONCERN .	6-63
6.8.1 Surface Water	6-63
6.8.2 Sediment	6-64
6.8.2.1 Organics	6-64
6.8.2.2 Inorganics	6-66
6.9 EXPOSURE ASSESSMENT	6-68
6.9.1 Contaminant Release, Migration, and Fate	6-69
6.9.2 Potential Exposure Pathways and Ecological	
Receptors	6-69
6.9.3 Assessment Endpoints	6-71
6.10 TOXICITY ASSESSMENT	6-71
6.10.1 Ecotoxicity Benchmarks	6-72
6.10.2 Toxicity Indices	6-81
6.11 RISK CHARACTERIZATION	6-82
6.11.1 Hazard Index (HI) Ratio	6-82
6.11.2 Summary of Risks and Uncertainties	6-82
6.11.3 Conclusions	6-83

Table of Contents (Cont.)

<u>Section</u>		<u>Page</u>
7	SUMMARY AND CONCLUSIONS	7-1
	7.1 SITE PHYSICAL HYDROGEOLOGY	7-1
	7.2 SOIL SAMPLING RESULTS	7-1
	7.3 GROUNDWATER MONITORING WELL SAMPLING RESULTS	7-2
	7.4 DRAINAGE NETWORK SAMPLING RESULTS	7-3
	7.5 FATE AND TRANSPORT	7-3
	7.6 HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENT	7-5
	7.6.1 Assessment of Human Health Risk	7-5
	7.6.2 Ecological Assessment	7-7
8	REFERENCES	8-1
<u>Appendix</u>		<u>Page</u>
A	PHASE I TECHNICAL MEMORANDA	A-1
VOLUME 2		
B	PHASE II TECHNICAL MEMORANDUM	B-1
C	PHASE III LEAD-SCREEN AUGER SAMPLING RESULTS	C-1
D	PHASE III SOIL BORING LOGS AND MONITORING WELL BORING LOGS	D-1
E	PHASE III ANALYTICAL RESULTS	E-1
F	PHASE I SOIL BORING LOGS AND MONITORING WELL BORING LOGS	F-1
G	CONTAMINANT MIGRATION MODELING METHODS	G-1
H	HUMAN HEALTH RISK TABLES	H-1

LIST OF TABLES

<u>Table</u>		<u>Page</u>
2-1	Analysis Program for Phase III Soil Boring Samples	2-11
2-2	Phase III Monitoring Well Specifications	2-14
2-3	Phase III Groundwater Elevation Data	2-15
2-4	Analysis Program for Phase III Groundwater Samples	2-22
2-5	Analysis Program for Surface Water and Sediment Samples	2-26
3-1	Grain-Size Distribution Results for Soil Samples Collected During the Installation of the Phase III Monitoring Wells	3-9
3-2	Phase III Vertical Hydraulic Gradients	3-10
3-3	Estimation of the Horizontal Flow Velocity and Travel Time for Groundwater	3-12
4-1	Summary of Organic Data for Sediment Samples	4-21
4-2	Summary of Inorganic Data for Sediment Samples	4-27
4-3	Summary of Organic Data for Surface Water Samples	4-31
4-4	Summary of Inorganic Data for Surface Water Samples	4-33
5-1	Physical and Chemical Properties of Chemicals of Potential Concern	5-16
5-2	Retardation Factors	5-17
5-3	Simplified Estimate of Chemical Mass Remaining in Aquifer	5-18
5-4	Groundwater Contaminant Mass Loading to the St. Joseph River	5-19

List of Tables (Cont.)

<u>Table</u>		<u>Page</u>
6-1	Adequacy of Detection Limits for Chemicals Detected at the Conrail Site	6-84
6-2	Data Qualifiers and Their Effect on Data Use in the Risk Assessment	6-87
6-3	Summary of Soil Boring Data (0.5 to 12 Feet BGS) for Organic Chemicals ($\mu\text{g}/\text{kg}$)	6-89
6-4	Summary of Groundwater Data for Organic Chemicals ($\mu\text{g}/\text{L}$)	6-90
6-5	Summary of Surface Water and Sediment Data for Organic Chemicals	6-92
6-6	Summary of Chemicals of Potential Concern for the Conrail Site	6-93
6-7	Summary of Potentially Complete Exposure Pathways Evaluated for the Conrail Site	6-94
6-8	Maximum and UCL Concentrations of Surface Soils (0.5-12 Feet BGS)	6-96
6-9	Facility Worker Exposure: Pathway 1A - Inhalation of Airborne Vapors	6-97
6-10	Residential Water Usage: Pathway 2A - Ingestion of Chemicals in Drinking Water (and Beverage Made Using Drinking Water) Adults	6-98
6-11	Residential Water Usage: Pathway 2B - Dermal Contact With Chemicals During Showering Adults	6-99
6-12	Residential Water Usage: Pathway 2C - Inhalation of Airborne (Vapor-Phase) Chemicals During Showering Adults	6-101

List of Tables (Cont.)

<u>Table</u>		<u>Page</u>
6-13	Residential Exposure: Pathway 2D - Inhalation of Indoor Airborne Vapors Adults	6-102
6-14	Future Scenario Facility Worker Exposure: Pathway 3A - Ingestion of Chemicals in Subsurface Soil	6-103
6-15	Future Scenario Facility Worker Exposure: Pathway 3B - Dermal Contact with Chemicals in Subsurface Soil	6-104
6-16	Future Scenario Facility Worker Exposure: Pathway 3C - Inhalation of Vapors From Newly Exposed Contaminated Subsurface Soils	6-105
6-17	Dermal Permeability Constants for Water and Dermal Absorption Coefficients for Soil Used in the Conrail Site Risk Assessment	6-106
6-18	EPA Weight-of-Evidence Categories for Chemical Carcinogenicity	6-107
6-19	Toxicity Values for Potential Carcinogenic Effects: Confidence in Classification and in Slope Factors (SFs)	6-108
6-20	Uncertainty Factors (Safety Factors) Used in the Derivation of Reference Doses	6-111
6-21	Toxicity Values for Potential Noncarcinogenic Effects	6-112
6-22	Summary of Estimated Excess Cancer Risks Associated with the Conrail Site	6-122
6-23	Summary of Estimated Hazard Indices for Noncarcinogenic Effects Associated with the Conrail Site	6-124
6-24	Special Species of Concern Located Within 2 Miles of the Conrail Site	6-126
6-25	Results of IDNR Fish Sampling Effort	6-127

List of Tables (Cont.)

Table		Page
6-26	Summary of Wetlands Surrounding Conrail Site	6-129
6-27	Screening of Surface Water Inorganic Results	6-130
6-28	Published Interim Sediment Quality Criteria	6-131
6-29	Calculated Sediment Quality Criteria	6-132
6-30	Screening of Sediment Samples	6-133
6-31	OME Provincial Sediment Quality Guidelines	6-136
6-32	T-Test of the Difference Between Background Sample and Downstream Sample Inorganic Analytical Results for Bangs Bay	6-137
6-33	T-Test of the Difference Between Background Sample and Downstream Sample Inorganic Analytical Results for the St. Joseph River	6-138
6-34	Summary of Hazard Indices for Aquatic Biota in Surface Water	6-139
6-35	Summary of Hazard Indices for Aquatic Biota in Sediment	6-135

List of Tables (Cont.)

Table		Page
6-13	Residential Exposure: Pathway 2D - Inhalation of Indoor Airborne Vapors Adults	6-102
6-14	Future Scenario Facility Worker Exposure: Pathway 3A - Ingestion of Chemicals in Subsurface Soil	6-103
6-15	Future Scenario Facility Worker Exposure: Pathway 3B - Dermal Contact with Chemicals in Subsurface Soil	6-104
6-16	Future Scenario Facility Worker Exposure: Pathway 3C - Inhalation of Vapors From Newly Exposed Contaminated Subsurface Soils	6-105
6-17	Dermal Permeability Constants for Water and Dermal Absorption Coefficients for Soil Used in the Conrail Site Risk Assessment	6-106
6-18	EPA Weight-of-Evidence Categories for Chemical Carcinogenicity	6-107
6-19	Toxicity Values for Potential Carcinogenic Effects: Confidence in Classification and in Slope Factors (SFs)	6-108
6-20	Uncertainty Factors (Safety Factors) Used in the Derivation of Reference Doses	6-111
6-21	Toxicity Values for Potential Noncarcinogenic Effects	6-112
6-22	Summary of Estimated Excess Cancer Risks Associated with the Conrail Site	6-122
6-23	Summary of Estimated Hazard Indices for Noncarcinogenic Effects Associated with the Conrail Site	6-124
6-24	Special Species of Concern Located Within 2 Miles of the Conrail Site	6-126
6-25	Results of IDNR Fish Sampling Effort	6-127

EXECUTIVE SUMMARY

This Remedial Investigation (RI) report of the Conrail Site has been completed by Ecology and Environment, Inc. (E & E) under contract to Region V of the United States Environmental Protection Agency (EPA) pursuant to Work Assignment No. 01-5L7Y, and various revisions. E & E planned and performed three phases of RI field activities detailed in this report. The primary objectives of the RI conducted by E & E were to assess the nature and extent of contaminated groundwater and identify contributing source areas, characterize the factors affecting the potential migration of contaminants, and evaluate the risk contaminants may pose to public health and the surrounding environment.

The Conrail Site study area is located approximately 1 mile southwest of the city of Elkhart, Indiana. The Conrail Site is composed of the Conrail railyard facility, light industrial property, commercial, residential, and agricultural areas and encompasses approximately 2,500 acres. The Conrail railyard facility is an active classification and distribution yard for freight cars and began operations in 1956 as part of the New York Central Railroad. The railyard continued operations as a subsidiary of the Penn Central Transportation Company (Penn Central) until April 1976, when Penn Central transferred its railroad operations to the Consolidated Rail Corporation (Conrail).

In June 1986, a resident in the County Road 1 area reported to EPA that elevated levels of volatile organic compounds (VOCs) were detected in his residential well. Based on this discovery, EPA/TAT initiated a groundwater sampling program in the County Road 1 and LaRue Street areas. Trichloroethene (TCE) concentrations as high as 4,870 µg/L and carbon tetrachloride (CCl₄) concentrations as high as 6,860 µg/L were detected. In response, bottled water and activated carbon filter units were provided by Indiana Department of Environmental Management (IDEM) to residents whose wells were affected.

EPA/TAT conducted a site assessment of the Conrail facility in July and August 1986. A total of seven water/liquid samples and 21 surface and subsurface soil samples were collected from various targeted locations of the Conrail facility on July 31 and August 1, 1986, respectively. Results of the analyses revealed TCE concentrations as high as 5,850 $\mu\text{g}/\text{kg}$, and CCl_4 concentrations as high as 117 $\mu\text{g}/\text{kg}$ in soil samples collected from the turnaround track area (Roy F. Weston 1986a). The Conrail facility was proposed for the National Priorities List (NPL) on June 24, 1988.

EPA issued a work assignment to E & E under the EPA Region V Alternative Remedial Contracting Strategy (ARCS) contract to conduct the RI/FS at this site. A summary of the RI results is provided below.

Site Physical Hydrogeology

The aquifer of concern is a glacial outwash deposit consisting of sand, and sand and gravel. The glacial outwash material is approximately 150 feet thick and lies on shale bedrock that is not a water-bearing unit. Depth to water varies between 3 and 20 feet below ground surface (BGS) in the study area. The general flow direction throughout the depth of the aquifer is to the west-northwest, except in the LaRue Street residential area, where the general flow direction is north. Hydraulic conductivity values derived from slug test data were used in addition to measured horizontal gradients to estimate the horizontal flow velocity of groundwater. Heterogeneity in site conditions cause variability in input parameters that result in a velocity range for groundwater of 11 feet per year to 2,200 feet per year. The mean horizontal flow velocity of groundwater is 200 feet per year.

Soil Sampling Results

A subsurface soil investigation was conducted as part of the field investigations in order to determine the nature and extent of identified and suspected source areas contributing to identified groundwater contamination. Based on analytical results from subsurface soil samples, two source areas on the Conrail facility have been identified.

A CCl_4 source area was identified in the eastern section of the classification yard near track 69 in the saturated zone. The dimensions of this source area are approximately 75 feet by 30 feet, with a vertical thickness of 7.5 feet. The analytical data from these subsurface

soil samples suggests that this CCl_4 source area extends beyond the boundaries established to date. CCl_4 contamination was also detected ($16 \mu\text{g/kg}$) in a 128 to 130-foot sample interval, but pooled residual CCl_4 was not detected on top of bedrock.

A TCE source area was identified in the western section of the classification yard between tracks 65 and 66. The approximate dimensions of this source area are 120 feet by 10 feet to a depth of 4 feet. This source area is in the unsaturated zone.

CCl_4 contamination was detected in the saturated zone in subsurface soil samples collected from the receiving yard on the Conrail facility, which is upgradient of the LaRue Street groundwater contamination area. CCl_4 concentrations were less than or equal to $31 \mu\text{g/kg}$. This contamination is a source of the identified, downgradient groundwater contamination in the LaRue Street area.

Groundwater Monitoring Well Sampling Results

CCl_4 and TCE contamination were detected in groundwater samples collected from monitoring wells screened in the shallow, intermediate, and deep zones on the Conrail facility, at the St. Joseph River, and in the area between these points, in the County Road 1 plume. Groundwater flow direction in all three zones is west-northwest, and the groundwater contamination plume extends continuously from the Conrail facility through Vistula Avenue and Charles Avenue to the St. Joseph River. Groundwater samples collected from monitoring wells located hydraulically upgradient of the plume and the site did not detect any VOC contamination.

The maximum concentration of CCl_4 groundwater contamination was $110,000 \mu\text{g/L}$ collected from monitoring well MW46S located in the track 69 source area. This concentration is 13.8% of the solubility of CCl_4 and suggests a CCl_4 dense non-aqueous phase liquid (DNAPL) source (EPA 1992a). Site background information and the detection of CCl_4 in a subsurface soil sample at 130 feet BGS also suggest that the CCl_4 source area in track 69 is a DNAPL source.

The maximum concentration of TCE detected in a groundwater sample was from monitoring well MW41, located immediately downgradient of the Conrail facility. This concentration is 1.4% of the solubility of TCE and suggests a TCE DNAPL source. MW41 is side-gradient of the TCE source area identified in the classification yard. Based on

analytical and hydrologic data, the DNAPL source is on the Conrail property, but is, as yet, unidentified.

CCl_4 was detected only in groundwater samples collected from shallow monitoring wells in the LaRue Street plume surrounding the identified CCl_4 soil contamination in the receiving yard on the Conrail facility. The soil contamination is thought to contribute to the identified CCl_4 groundwater contamination. Monitoring wells upgradient of the soil contamination did not detect CCl_4 .

TCE was detected only once (MW20S - 15 $\mu\text{g/L}$) above the MCL for TCE in the LaRue Street plume. TCE was detected below the MCL throughout the plume. Monitoring wells located upgradient of the Conrail facility did not detect TCE. Based on groundwater analytical data, the source of the TCE contamination is on the Conrail property, but is, as yet, unidentified.

Drainage Network Sampling Results

Analysis of the sediment and water samples from the Conrail drainage network revealed no detectable levels of TCE or CCl_4 . The data suggest that the drainage network is not currently a source of the identified groundwater contamination, which resulted from surface spills on the facility. The possibility that the drainage network historically acted as a conduit for VOC contamination cannot be assessed due to lack of previous sampling data from the network.

Fate and Transport

A focused discussion on the subsurface and groundwater fate and transport of CCl_4 and TCE is presented. Processes such as volatilization, liquid transport, sorption, and transformation reactions have likely occurred at the site based upon the contaminants present and the observed environmental conditions. Volatilization of the dissolved chlorinated aliphatic hydrocarbons from the groundwater that is present at or near the water table can transfer significant contaminant mass from groundwater to soil gas. There is a high probability that this process has occurred and is currently operative in the study area. Liquid transport is occurring at the site as dissolved contaminants in groundwater undergo migration. If DNAPL is present, it may also migrate by density-driven liquid transport processes specific

to DNAPLs. Sorption of chemicals of potential concern onto aquifer materials is expected to be an important process at the site. Analytical results of total organic carbon in the aquifer material are combined with chemical-specific data to quantify sorption so that retardation can be estimated. The retardation is used to estimate the migration rate of contamination relative to the groundwater flow velocity. Of the numerous transformation reactions that may possibly occur, sequential reductive dehalogenation is important because it appears to be functioning at the site due to detection of transformation products.

The fate and transport processes coupled with site-specific data enables the estimation of movement, mass, and loading of CCl_4 and TCE. Due to the heterogeneity in site conditions, it is estimated to take from 6 to 1,200 years for the groundwater to travel from source to the St. Joseph River. If retardation does not take place, contamination would undergo advection at the same rate as groundwater flow. If available sorptive capacity in the aquifer materials permits sorption, it is estimated that CCl_4 and TCE will travel at approximately 40 percent of the rate of groundwater. The total mass of CCl_4 and TCE remaining in the aquifer as dissolved contaminants in the groundwater and sorbed to aquifer materials can be estimated based on analytical data from the site and an estimate of DNAPL volume. The estimated total mass of CCl_4 and TCE in the groundwater and sorbed to aquifer materials is 20,000 pounds. Residual DNAPL may contribute 150,000 pounds as CCl_4 and TCE, combined. The estimated loading from the site to the St. Joseph River is 20 pounds of TCE and 20 pounds of CCl_4 per year. Assuming this loading, it would take 200 years to remove the CCl_4 and TCE that is dissolved in the groundwater presently in the aquifer. If DNAPL is present, this estimate of elapsed time for natural attenuation would be much greater.

Transformation reactions resulting in the formation of daughter products of CCl_4 and TCE are occurring at the site. The daughter compounds chloroform and 1,2-dichloroethene were detected in monitoring well samples that also contained higher concentrations of CCl_4 and TCE, respectively.

Human Health and Ecological Risk Assessment

Assessment of Human Health Risk

The human health assessment examined the risks to railyard workers and residents of nearby neighborhoods from exposures to Conrail Site contaminants. The seventeen chemicals selected as contaminants of potential concern were all volatile organic compounds.

Potential risks to railyard workers were evaluated for existing conditions and for a possible future excavation scenario. Only the vapor inhalation pathway was evaluated under the current risk scenario. The potential excess cancer risk for the reasonable maximum exposure (RME) case for this pathway is 3.40×10^{-3} , which exceeds EPA's threshold value range (10^{-6} to 10^{-4}) for upper-bound lifetime cancer risks to an individual. The hazard index for the RME case for this pathway is 253, which indicates that there is potential for adverse noncarcinogenic risks via this pathway.

The potential future risks to railyard workers were evaluated via the soil contact pathway (direct contact and incidental ingestion) and inhalation of airborne contaminants. These pathways were evaluated for a possible excavation scenario, in which it is envisioned that current subsurface contamination is exposed and then becomes surface contamination. The potential excess cancer risks and the potential adverse noncarcinogenic risk calculated for the soil contact pathway exceed their associated EPA threshold values. This indicates that there is a potential for carcinogenic and noncarcinogenic risks to workers during excavation activities.

Two distinct plumes of groundwater contamination, designated as Plume 1 (County Road 1 Plume) and Plume 2 (LaRue Street Plume), respectively, are migrating from the railyard towards the St. Joseph River. Plume 1 is migrating north and west of the facility, while Plume 2 is migrating north and east of the facility. Plume 1 contained 10 COPCs while Plume 2 contained seven. The potential risks posed by each plume varied based not only on contaminant concentration, but also based on the toxicological properties of the chemicals detected in the plumes, and on the pathway of exposure.

Residential exposures from groundwater usage and inhalation of indoor airborne contaminants from groundwater vapors was evaluated for both plumes. The residential groundwater usage pathway included ingestion of drinking water, dermal contact with water during showers, and inhalation of water vapors during showers. The inhalation of indoor

airborne contaminants from groundwater vapors were evaluated based on an empirical model of groundwater contaminants volatilizing into nearby unsaturated soil gas, infiltrating into basements and subsequently diffusing throughout the entire home.

The potential risks from these exposures were greater in Plume 1. The excess cancer risk for the RME case of groundwater usage is 9.00×10^{-3} . The excess cancer risk associated with inhalation of indoor airborne contaminants for the RME case was 1.71×10^{-4} . These values all exceed the EPA threshold value range of 10^{-6} to 10^{-4} . The potential for noncarcinogenic adverse risks from residential exposures to contaminants in Plume 1 were also above the EPA hazard index threshold level of 1 for the groundwater usage pathway (hazard index = 201). The potential cancer risks in Plume 1 are due to exposure to CCl_4 , TCE, chloroform, 1,1-DCE, and vinyl chloride. The potential adverse noncarcinogenic risks due to Plume 1 groundwater usage is due solely to carbon tetrachloride.

Residential exposures to Plume 2 contaminants results in potential excess cancer risks from both groundwater usage and inhalation of airborne contaminants from groundwater vapors. The RME case risk via the groundwater usage pathway is 1.45×10^{-3} . The RME case risk via the indoor air inhalation pathway is 1.11×10^{-5} .

The hazard index for the Plume 2 groundwater usage pathway is 4.16. This indicates that there is a potential for long-term, chronic adverse noncarcinogenic risks to individuals.

The excess cancer risks for Plume 2 groundwater usage and indoor air inhalation pathways are due to CCl_4 , chloroform, benzene, and TCE. The noncarcinogenic risks in Plume 2 are due solely to carbon tetrachloride.

Ecological Assessment

The ecological assessment examined existing and potential risks posed by site-derived contaminants to nearby natural habitats and associated flora and fauna. The primary focus of the risk assessment was on aquatic organisms in Baugo Bay, the St. Joseph River, and the Conrail ponds.

Four special species of concern were identified within 2 miles of the Conrail Site. Two were plant species whose latest sightings were in the 1930s and the 1940s. One mammal, the badger (*Taxidea taxus*), has been sighted recently north of the St. Joseph River, which is an area that is probably not affected by the site. The Cooper's Hawk (*Accipiter*

cooperii) has been recently sighted near the site. However, the hawk prefers to nest in mature trees, will not hunt within 0.5 miles of the nest, and will travel up to 2 miles to hunt over open fields and along woodland margins. Therefore, it is unlikely that site-related contaminants will effect the Cooper's Hawk.

Surface water CPECs were chosen based on comparisons of surface water sample analytical results with current ambient water quality criteria. Lead was chosen as a surface water CPEC in the St. Joseph River, while lead and zinc are CPECs for the ponds. No surface water CPECs were selected for Baugo Bay.

Organic CPECs in sediments were chosen based on comparisons of sediment sample analytical results with sediment quality criteria. For those chemicals that did not have published values, the criteria were calculated based on the equilibrium-partitioning approach for non-polar organics. No organic CPECs were chosen for Baugo Bay or for the St. Joseph River. The five CPECs chosen for the ponds are 4,4'-DDT, 4,4'-DDD, 4-methylphenol, fluorene, and Aroclor 1254.

Many inorganic contaminants were detected in Baugo Bay, the St. Joseph River, and in the ponds. A statistical comparison of the inorganic results from the background locations and the results from the downstream locations for Baugo Bay and the St. Joseph River indicates that there was no significant difference between the concentrations of contaminants in the two locations. Therefore, the contamination does not appear to be site-related. There were no background ponds available for similar statistical comparisons for the pond data. Therefore, CPECs chosen for the pond area are arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, and zinc.

Sediment hazard indices (the ratio of the sample concentration to the toxicity reference value) were greater than 1 for cadmium, copper, lead, zinc, fluorene, and 4-methylphenol. Arsenic, chromium, manganese, mercury, nickel, DDT, DDD, and Aroclor 1254 also had indices greater than 1, but the magnitudes of the indices were lower than for the previously stated CPECs. The maximum concentrations of these CPECs were all located in the westernmost pond.

The surface water hazard index for lead is greater than 1 while the hazard index for zinc is less than 1. As in the sediment CPECs discussed above, the highest concentrations of surface water CPECs were found in the westernmost pond.

Potential risks to benthic organisms from CPECs present in the ponds are assessed to be moderate to high. CPECs were detected only in the westernmost pond, which receives drainage from an outfall of the facility drainage system. This pond is less than 1 acre, and other suitable habitats are available for use in the area. Non-benthic organisms that use the pond for a portion of their life cycle may be potentially impacted by the sediments, but they are at a lower risk than benthic organisms. The terrestrial ecosystem in the general vicinity of the site does not appear to be impacted by site-related CPECs.

1. INTRODUCTION AND SITE BACKGROUND

This Remedial Investigation (RI) report was prepared for the United States Environmental Protection Agency (EPA) pursuant to Work Assignment No. 01-5L7Y, and various revisions, issued to Ecology and Environment, Inc. (E & E), under the Region V Alternative Remedial Contracting Strategy (ARCS) Contract No. 68-W8-0086. Under this work assignment, E & E was directed to conduct a Remedial Investigation and Feasibility Study (RI/FS) at the Conrail Site, located in Elkhart, Indiana. The RI was conducted in three phases. The objectives of each investigative phase of the RI are detailed in the Work Plan for that particular phase (E & E 1989a, 1991a, 1992b, respectively). This report presents and summarizes the findings of the Phase III RI activities conducted in the study area and interprets findings from all three phases of the RI. Documents detailing the results of the Phase I and II investigations are included as appendices to this report.

The information obtained from the three field investigations was used to determine the nature and extent of contamination as well as the fate and transport processes that influence the observed contaminant distribution. The field investigation data provide the basis for assessment of the risk on human health and the environment posed by the observed contaminant distribution. The field investigations also provide data necessary to develop and to evaluate remedial alternatives during the Feasibility Study (FS).

This report consists of eight sections. Section 1 presents introductory and background information for the Conrail Site. Section 2 presents a description of the procedures employed for the Phase III field activities and includes tabulated data concerning the physical specifications of monitoring wells, sample numbers, and sample types. Section 3 presents the Phase III physical results related to the geology of the study area and contains a comprehensive geologic and hydrogeologic analysis. The Phase III chemical analytical results are summa-

ized in Section 4, which also includes a discussion of the nature and extent of contamination. Section 5 includes an interpretation of the observed contaminant distribution and anticipated migration using analytical groundwater modeling and principles of subsurface fate and transport. Both components of the risk assessment, the human health evaluation (HHE) and the ecological assessment (EA), are included in Section 6. Section 7 presents a summary of the investigation finding and conclusions. References used to develop the RI report are listed in Section 8.

1.1 SITE DESCRIPTION

The Conrail Site study area is located approximately 1 mile southwest of the city of Elkhart, Indiana (see Figure 1-1). The study area encompasses approximately 2,500 acres and lies in portions of both Elkhart and St. Joseph counties. The study area is bordered on the north by the St. Joseph River, on the west by Baugo Bay and Baugo Creek, on the south by the southern property line of the Conrail facility, and on the east by Nappanee Street. The 675-acre Conrail railyard facility comprises the southern portion of the study area south of U.S. 33. Several light-industrial properties are located north of U.S. 33, to the north of the Conrail railyard. The study area also includes residential areas situated south of the St. Joseph River and north of U.S. 33. Groundwater contamination has been identified in these residential areas, designated as the County Road 1, LaRue Street, Vistula Avenue, and Charles Avenue areas.

1.2 SITE PHYSICAL FEATURES

The Conrail Site study area is located in the floodplain of the St. Joseph River. Topography in the study area is relatively flat, with ground surface elevations varying from 760 to 725 feet above mean sea level (MSL). Elevations are generally highest in the Conrail facility. Elevations decrease as the ground surface slopes towards the St. Joseph River. The St. Joseph River is a mature, meandering river, which forms the northern border of the study area. The stage of the St. Joseph River, as measured by E & E at the Ash Road bridge as it crosses the river, ranges between 717 and 715 feet above MSL and the flow of the river is regulated by a hydroelectric plant located in Elkhart, Indiana, upstream of the study area.

Surface soils at the Conrail facility have been disturbed and no series is recognizable due to the presence of railroad ballast. Sixty-five percent of the surface material in the Conrail facility is composed of stone ballast and concrete (Jacobs 1987). Several features important in the daily operations of the Conrail facility are present (see Figure 1-2, Site Features Map). The largest feature is the main classification yard, which consists of 72 tracks that are sequentially numbered from track 1 in the northernmost area of the classification yard to track 72 in the southernmost area of the yard. The 72 tracks are divided into eight groups of nine tracks per group. Rail cars are sorted and redistributed by decoupling trains at the hump tower and, through gravity-driven coasting, subsequently recoupling cars by collision once they are directed to the appropriate track. Three ponds are located south of the classification yard on the Conrail facility. The hump tower is located east of the main classification yard. The receiving yard is a collection of tracks that extends from the hump tower eastward toward and past Nappanee Street. Another set of tracks north of the main classification yard contains a main line that handles through-traffic. A turnaround track is present between the main classification yard and the receiving yard. A diesel shop is located northeast of the main classification yard and west of the turnaround track. A car shop is located in the west end of the main classification yard, near its north-south center point. The Bridges and Buildings (B and B) Shop is located west and slightly south of the car shop.

1.3 LAND USE AND POPULATION

Land use in the study area is diverse. Portions of the study area that are not occupied by railroad operations consist of residential, light-industrial, and agricultural areas. Small commercial establishments occupy some localized areas on U.S. 33 and on Ash Road. The four residential areas of concern mentioned previously, County Road 1, Vistula Avenue, Charles Avenue, and LaRue Street, are present in the study area (Figure 1-1). County Road 1 residential area is located in the appropriate vicinity of the intersection of County Road 1 and U.S. 33. Vistula Avenue and Charles Avenue residential areas are located near the St. Joseph River and Bungo Bay, and Ash Road. The LaRue Street residential area is located east in the study area and north of U.S. 33. The estimated populations in these residential areas that are potentially impacted by site related groundwater contamination are 371, 258,

381, and 650, respectively (E & E 1991c). The residents in these four areas utilize private wells.

1.4 GROUNDWATER RESOURCES, SURFACE WATER RESOURCES, AND CLIMATE

The Elkhart area is underlain by a regionally extensive, thick glacial outwash deposit composed of unstratified sand, and sand and gravel, which serves as the aquifer for drinking water in the area. The aquifer is classified IIA and is a nationally designated sole source aquifer. The sand deposit has an average hydraulic conductivity value of 80 ft/day and the sand and gravel deposit has an average value of 400 ft/day (Imbrigiotta and Martin 1981). Bedrock aquifers are not considered an important source of water in this area due to their depth, relatively low-yielding character, and the presence of a productive aquifer in the glacial material (Imbrigiotta and Martin 1981). The bedrock below the unconsolidated glacial material is an approximately 170-foot thick sequence of shale.

Groundwater flow data indicate that the St. Joseph River is hydraulically connected to the outwash aquifer in the study area and is a discharge zone for this aquifer (Imbrigiotta and Martin 1981). Groundwater pumpage is another discharge source for this aquifer. The pumping location of the predominant groundwater user in the area, Suburban Utilities, is shown in Figure 1-1. The Suburban Utilities well field consists of two 10-inch-diameter wells that are each 90 feet deep. According to the pumping schedule, each well is operational in alternating months, thus allowing only one of the wells to be pumped at any given time. One of the wells produces 160 gallons per minute, the other 200 gallons per minute. The municipal wells are used to supply drinking water to residences east of the study area. Private wells are used to supply the drinking water in the residential areas of the site. The city of Elkhart's municipal supply wells are located approximately 3 miles northeast of the site. Precipitation and private septic systems are the only known sources of recharge to this outwash aquifer.

The major surface water bodies in the vicinity of the study area are the St. Joseph River and Baugo Bay. The St. Joseph River flows to the west and is located approximately ½ mile north of the Conrail facility. Baugo Bay drains north into the St. Joseph River, and is located approximately ½ mile west and north of the Conrail facility. Baugo Creek, which

forms part of the southeast boundary of the study area, flows north into Bangs Bay. Crawford Ditch originates at the northern border of the railyard and flows northward intermittently to the St. Joseph River. Three elongated ponds, classified as wetlands (Section 6), are present along the southern border of the Conrail facility. These surface water resources are discussed in detail in the Ecological Assessment (EA) (Section 6.6).

Elkhart County has a continental climate that is essentially uniform over the entire county. The average precipitation is 35.8 inches per year, and the average annual temperature is 49.5° F (Crompton *et al.* 1986). For the period 1951 to 1980, the average monthly rainfall ranged from a low of 1.8 inches in February to a high of 3.8 inches in August (Crompton *et al.* 1986). Elkhart County has four well-defined seasons. July is the warmest month of the year, with an average monthly temperature of 74.0° F. Temperatures of 90° F or higher are recorded on an average of eight days per year. Relative humidity on an average summer day varies from 40 percent in the afternoon to 90 percent or higher just before dawn. Fall is usually marked by moderate temperatures and by sunshine, which averages approximately 70 percent of the daylight hours. In winter, there is an average of seven days with temperatures below 0° F. January is usually the coldest month of the year with an average temperature of 23.1° F (Crompton *et al.* 1986). Snowfall occurs as early as October and as late as May. The largest amount of snowfall occurs in February, and the average yearly snowfall is 25.9 inches. The soil is typically frozen for three to four months during the winter (U.S. Department of Agriculture Soil Conservation Service 1974).

1.5 SITE HISTORY AND PREVIOUS INVESTIGATIONS

In 1956, the railyard began operations as part of the New York Central Railroad; later, the railyard continued operations as a subsidiary of the Penn Central Transportation Company (Penn Central). In April 1976, Penn Central transferred its railroad operations to the Consolidated Rail Corporation (Conrail). In October 1978, Penn Central finalized a reorganization plan that transferred all of its rail assets to Conrail. The Conrail facility currently serves as a classification and distribution yard for freight cars and is the primary connection between the Chicago area and Conrail's Northeastern rail system. The railyard also operates car repair, car cleaning, and diesel refueling facilities on site (Jacobs 1987). An interview conducted by Indiana Department of Environmental Management (IDEM) personnel

with a former Conrail employee revealed that solvents were used at the car shop and diesel shop for degreasing purposes. Site background information indicates that a spill of carbon tetrachloride (CCl_4) occurred in the area of track 69 in the main classification yard of the Conrail railyard.

In June 1986, a resident in the County Road 1 area reported to EPA that elevated levels of volatile organic compounds (VOCs) were detected in his residential well. On July 2, 1986, EPA's Technical Assistance Team (TAT) collected and analyzed a water sample from this residential well. Sample analysis indicated the presence of trichloroethylene (TCE) at 800 $\mu\text{g/L}$ and CCl_4 at 485 $\mu\text{g/L}$. Based on this discovery, EPA/TAT initiated a groundwater sampling program in the County Road 1 and LaRue Street areas. Groundwater sampling began on July 17, 1986. A total of 88 residential wells was sampled by EPA/TAT. Concurrently, an additional 11 residential wells were sampled by individual well owners. TCE concentrations as high as 4,870 $\mu\text{g/L}$ and CCl_4 concentrations as high as 6,860 $\mu\text{g/L}$ were detected in this round of sampling. CCl_4 or TCE were detected in a total of 63 groundwater samples. Thirty-two groundwater samples had compound levels above the 10-day health advisories for either TCE or CCl_4 (128 $\mu\text{g/L}$ for TCE and 12 $\mu\text{g/L}$ for CCl_4) (Roy F. Weston 1986b).

Bottled water was provided by IDEM to residents whose wells were affected. Additionally, two types of activated carbon filter units were installed in residences: point-of-use units and whole-house units. A total of 20 point-of-use and 56 whole-house units were installed (Roy F. Weston 1986b). IDEM was responsible for operation and maintenance of these units until 1992, when Groundwater Technology, Inc. (GTI), of Indianapolis, Indiana, took over this responsibility on behalf of Conrail, as part of a Unilateral Administrative Order (UAO) issued by EPA for an Interim Remedial Action at the site.

EPA/TAT conducted a site assessment of the Conrail facility in July and August 1986. A total of seven water/liquid samples and 21 surface and subsurface soil samples were collected from various targeted locations of the Conrail facility on July 31 and August 1, 1986, respectively. Results of the analyses revealed TCE concentrations as high as 5,850 $\mu\text{g/kg}$, and CCl_4 concentrations as high as 117 $\mu\text{g/kg}$ in soil samples collected from the turnaround track area (Roy F. Weston 1986a).

The EPA Environmental Response Team (ERT) conducted a soil-gas survey on September 2 and September 4, 1986, along the northern perimeter of the Conrail facility in the LaRue Street and County Road 1 areas. No conclusive data were obtained from the survey (Roy F. Weston 1986c).

Peerless-Midwest, Inc., contracted by a private development company, studied the water quality and hydrology in the vicinity of the Charles Avenue area. The study was conducted prior to residential development because of reported groundwater contamination in the area. As part of this study, six monitoring wells were installed; these monitoring wells and one existing residential well were utilized in the study. A pump test was also conducted. Groundwater samples from the shallow wells (< 30 feet deep) did not reveal the presence of TCE or CCl_4 , but samples from the deep monitoring wells (> 110 feet deep) were all contaminated with TCE and CCl_4 (with maximum values of 2,495 $\mu\text{g/L}$ and 388 $\mu\text{g/L}$, respectively) (Peerless-Midwest, Inc. [no date]).

EPA issued a work assignment to Jacobs Engineering Group, Inc., to perform a Potentially Responsible Party (PRP) search for the Conrail facility. The results of this search were presented in a three-volume report (Jacobs Engineering Group, Inc. 1987). The Conrail facility was proposed for the National Priorities List (NPL) on June 24, 1988.

Subsequently, EPA issued a work assignment to E & E under the EPA Region V Alternative Remedial Contracting Strategy (ARCS) contract to conduct the RI/FS at this site. Beginning in July 1989, E & E conducted a Phase I RI at the Conrail site. The results of the Phase I investigation were presented and discussed in the following technical memoranda, which are included in Appendix A: *Technical Memorandum of Soil Gas Sampling*, November 1989 (E & E 1989c), *Technical Memorandum of Groundwater Monitoring and Sampling*, April 1990 (E & E 1990a), *Preliminary Evaluation of Phase I RI Results and Interim Remedial Alternatives*, April 1990 (E & E 1990b), and *Technical Memorandum of Slug Test Results* (E & E 1990c). Following an evaluation of the data collected during the Phase I RI, E & E recommended, and EPA concurred, that a second phase of investigation be conducted to address project objectives. E & E completed a *Phased Feasibility Study Report for the Conrail Site* (PFS), including Phase I results, in April 1991 (E & E 1991c). A Record of Decision (ROD) for interim groundwater remedial action at the Conrail Site was signed in June 1991, selecting a remedy that followed the findings presented in the PFS.

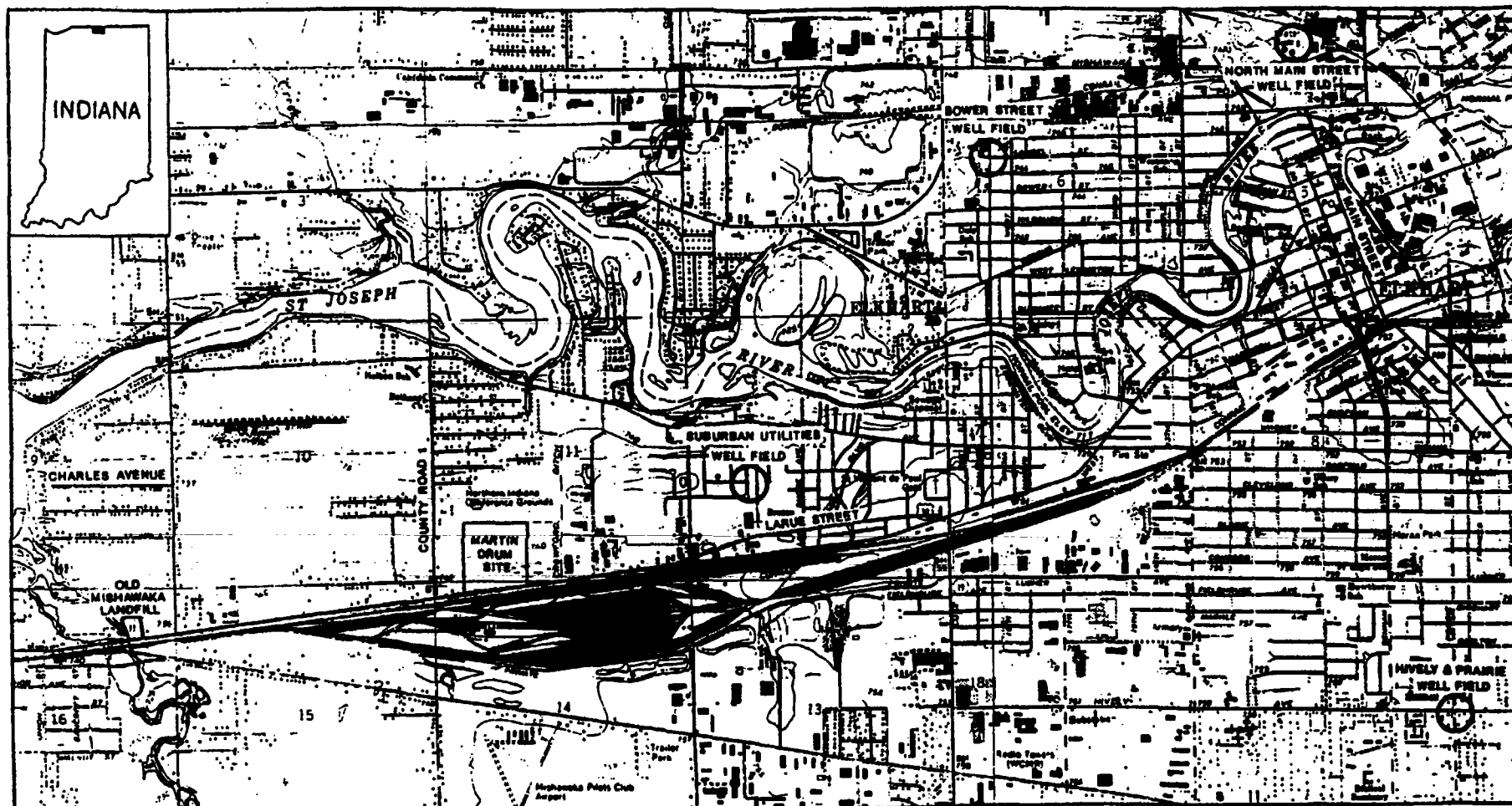
In July 1991, E & E initiated the Phase II RI at the Conrail Site. In July 1992, E & E submitted the *Phase II Remedial Investigation, Conrail RI/FS, Technical Memorandum* (E & E 1992a) (Appendix B) to EPA. The *Phase II Technical Memorandum* summarized, integrated, and presented interpretations and conclusions based upon data gathered during the Phase I and Phase II field investigations.

On July 7, 1992, EPA issued a UAO for Remedial Design and Remedial Action, which binds Conrail and Penn Central to perform remedial activities described in the ROD and Statement of Work (SOW) attached to the Order. The interim remedial action for the Conrail Site, as described in the SOW, will consist of the following elements:

- Institutional Controls including deed restrictions for future use of the railyard executed through the Elkhart County Recorder; restrictive covenants ensuring that property outside the Conrail railyard on which components of the remedy will be located (e.g., monitoring wells, treatment facilities) will not be disturbed; and abandonment of residential wells located within the area of contamination;
- Monitoring Program including groundwater monitoring in and around the area of contamination and air monitoring of the treatment system;
- Groundwater Extraction, Collection, Treatment, and Discharge System will be designed, constructed, operated, and maintained to prevent further horizontal and vertical migration of contaminated groundwater located northwest, downgradient from the Conrail railyard by extracting water from the plume, treating it using air stripping, and discharging it to the St. Joseph River;
- Fence Installation to enclose groundwater extraction and treatment facilities;
- Provision of an Alternate Water Supply through the design, construction, and first-year operation and maintenance of a distribution system extending from the City of Elkhart water supply to affected residential/business areas located downgradient from the Conrail railyard, and maintenance of individual water filter units or provision of bottled water for those areas until the distribution system is operational.

Conrail has retained GTI to design and implement the interim groundwater remedial actions outlined in the SOW.

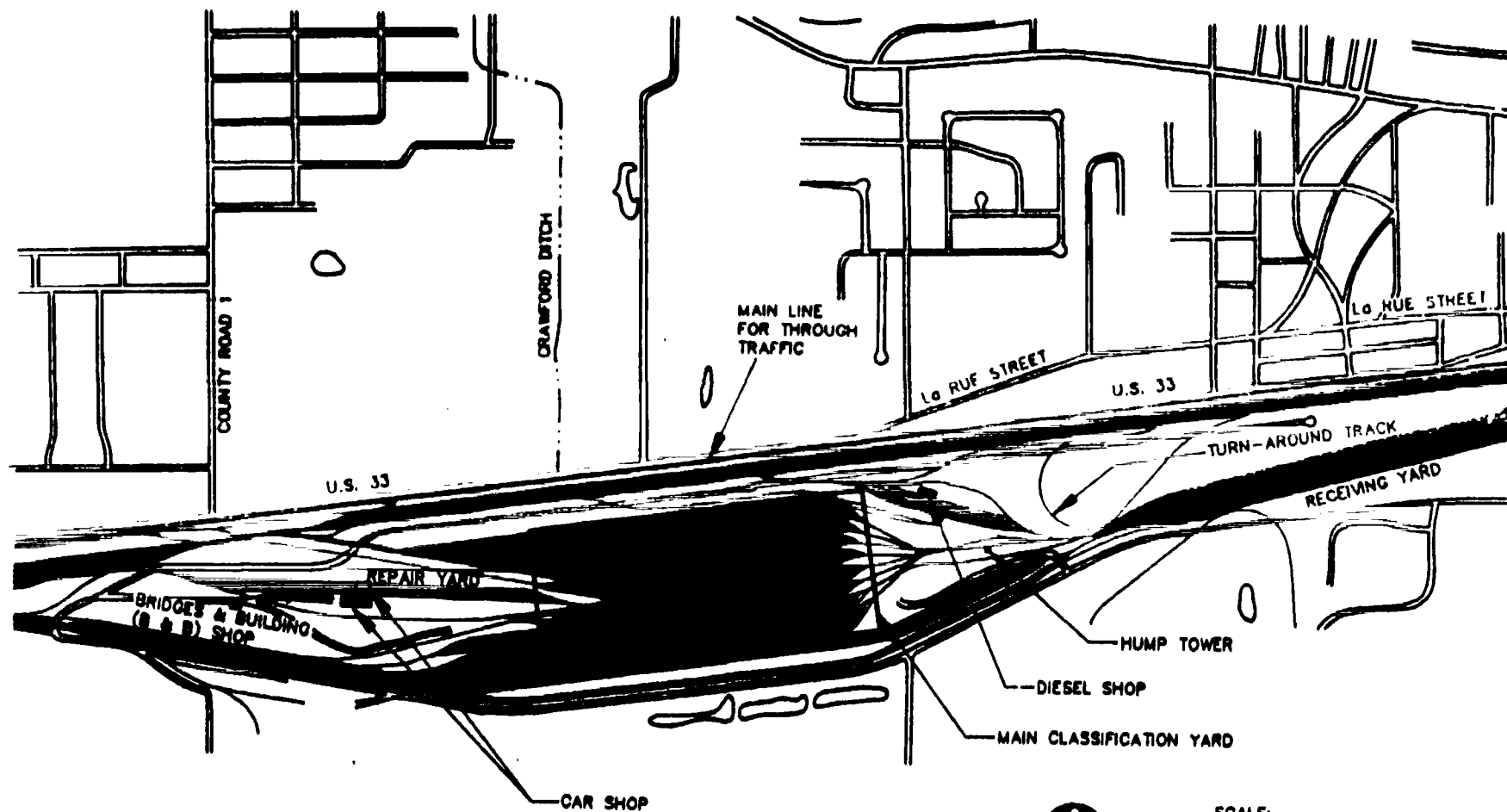
Following submittal of the *Phase II Technical Memorandum* in July 1992, E & E recommended, and EPA concurred, that a third phase of investigation be conducted to meet the remaining data requirements needed to address the RI objectives. In November 1992, the *Comrail RI/FS Phase III Work Plan* (E & E 1992b) was submitted to EPA. The Phase III field investigation began in late November, 1992, and was concluded by early February, 1993.



SOURCE: Ecology and Environment, Inc., 1982; BASE MAPS: USGS, Ekhart, IN Quadrangle, 7.5 Minute Series, 1961, Photorevised 1981; USGS, Coonata, IN Quadrangle, 7.5 Minute Series, 1960, Photorevised 1980.

SCALE
0 1/2 1000 FEET

FIGURE 1-1 STUDY AREA LOCATION MAP



SCALE:
1" = 1200'

FIGURE 1-2
SITE FEATURES MAP

2. PHASE III FIELD INVESTIGATION PROCEDURES

This section presents the procedures used by E & E for data collection during the Phase III field investigation.

2.1 LEAD-SCREEN AUGER SAMPLING

From December 1, 1992, to January 26, 1993, E & E subcontracted Bergerson-Caswell, Inc. (BC), of Maple Plain, Minnesota, to perform drilling and related activities for the Conrail Phase III field investigation. The first part of this work, from December 1 to December 23, 1992, consisted of lead-screen auger (LSA) drilling and groundwater sampling activities. Use of the LSA technique allowed sampling of the groundwater within the plume(s) at discrete depth intervals to determine the vertical extent of contamination, and to provide information on the optimum depths for monitoring well screened intervals. The technique also was used to investigate the location and extent of potential source areas. A total of 16 LSA borings was completed by BC under the direction of E & E. The locations of the Phase III LSA borings, LSA31 through LSA46, are shown in Figure 2-1. Samples analyzed at the field laboratory included 310 groundwater samples from discrete lead-screen intervals, together with 135 duplicate samples, field blanks, and trip blanks. Samples were analyzed using the purge and trap method with a gas chromatograph (GC) for CCl_4 , chloroform (CHL), (TCE), and 1,1,1-trichloroethane (TCA). These compounds were chosen based on the results from the Phase I and II investigations.

The LSA sampling was in accordance with the procedures specified in the Field Sampling Plan (FSP). At each LSA boring location, BC used either a CME 75 or a Gus Pech drill rig to advance 4-1/4-inch inside diameter (ID) hollow-stem augers, coupled with a slotted lead auger, by conventional hollow-stem auger drilling methods. A stainless steel plug was

inserted into the end of the lead auger to prevent heaving sands from entering the auger. The augers were advanced at 5-, 10-, or 20-foot depth intervals. At these discrete depths, the 5-foot LSA was sealed from the flights above it with a downhole packer system. The packer system was constructed of a sliding head inflatable packer that expanded radially as pneumatic pressure was applied. The expandable packer gland was mounted on a 2-inch ID mandrel pipe. This packer assembly was attached to the end of a 2-inch ID stainless steel riser pipe and positioned downhole just above the LSA. When in position, the packer was inflated with nitrogen, creating an effective seal over the entire length of the packer gland element. This seal isolated the LSA and thus minimized the volume of purge water generated. With the packer inflated, the level of standing water in the riser was measured with a graduated stainless steel chalked tape so that the volume of water in the LSA section plus the 2-inch riser could be calculated. A minimum of three standing volumes of water was purged from the LSA and riser with either a 1.75-inch outside diameter (OD) Keck™ or Grundfos™ variable flow submersible pump positioned inside the LSA. Between volumes, the purge water was tested for pH, conductivity, and temperature. Purging was considered complete when all three parameters had stabilized for three consecutive readings (± 0.25 pH units, ± 50 μ mhos/cm, and ± 0.5 °F, respectively). If the LSA plus riser pumped dry, 15 minutes of recharge was allowed and then pumping was resumed. If the LSA plus riser pumped dry three times, the interval was considered to be purged and the sample was collected as soon as sufficient recharge had occurred. Following purging, the flow rate of the submersible pump was substantially reduced and the groundwater sample was collected directly from the pump's discharge hose into two 40-mL glass volatile organic analysis (VOA) vials with zero headspace. Each vial was labeled with boring number, depth of sample, and date/time of collection. Samples were cooled immediately on ice and transported to the field laboratory for CCl_4 , CHL, TCE, and TCA analyses. A daily trip blank was prepared by the field chemist from deionized water, transported to the site, and analyzed at the end of each day.

After each sample was collected, the pump, packer, and riser were removed from the augers and decontaminated with a steam cleaner. In addition, a solution of Liquinox™ and potable water, followed by a potable water rinse, was run through the pump between samples to decontaminate the internal elements of the pump and discharge hose. At least one rinse blank sample of potable water run through the pump was collected daily and analyzed at the

field laboratory to monitor internal decontamination of the pump. In addition, samples collected directly from the potable water source were analyzed occasionally throughout the LSA program to ensure that this water remained free from contamination. The source of potable water was the City of Elkhart Water Works municipal supply.

Following completion of each LSA boring, the stainless steel plug was removed from the auger and the borehole abandoned by tremie-sealing with Enviropug™ bentonite grout as the augers were withdrawn. Drill cuttings were sealed in 55-gallon drums and bulked later in roll-off boxes. All purge and pump decontamination water was also collected for disposal (see Section 2.7, Investigation-Derived Waste).

2.2 SOIL SAMPLING

Soil sample collection procedures were performed in accordance with the FSP. From January 18 to January 25, 1993, 13 soil borings, B40 through B52, ranging in depth from 1.5 feet to 150 feet below ground surface (BGS) were completed. Five distinct areas were targeted in the Phase III subsurface soil investigation. Figure 2-2 presents the locations of the Phase III soil borings. Two of the areas were sampled to further define the extent of source areas identified during the Phase II field investigation: track 69 area in the eastern end of the classification yard and track 65 and 66 area in the western end of the classification yard. All soil samples collected from these 2 areas were analyzed only for Target Compound List (TCL) volatile organic compounds (VOCs) at a Contract Laboratory Program (CLP) laboratory. Three additional areas were investigated to determine if they were potential source areas contributing to known groundwater contamination: an area in the receiving yard upgradient to LaRue Street area groundwater contamination, an area of reported spilled drums located south of the repair yard and east of the car shop, and a location of a reported buried tank car. Soil samples collected from these 3 areas were analyzed for TCL VOCs, semivolatiles, and pesticides and PCBs, and Target Analyte List (TAL) metals at a CLP laboratory.

Drilling was conducted by BC with the CME 75 rig and 4-1/4-inch ID hollow-stem augers. Soil samples were collected continuously or every 2.5 feet with a 3-inch OD, 24-inch-long split-spoon. Each split-spoon soil sample was visually classified by an E & E geologist, who maintained a detailed log of sample depth, blow count, recovery, soil description, and organic vapor analyzer (OVA) readings. Phase III soil boring logs are

presented in Appendix D. Between samples, the split-spoons were decontaminated with Liquinox™ solution, isopropanol, and distilled water. Between borings, the augers were decontaminated with a steam cleaner. After each boring, the hole was tremie-grouted to the surface with Enviroplug™ bentonite grout as the augers were withdrawn. Drill cuttings were contained in 55-gallon drums and later bulked in roll-off boxes for future disposal.

The OVA screening results, visual observations, and LSA results, where applicable, were the primary criteria for selection of three depth-specific samples from each soil boring for submittal to CLP laboratories for VOC analysis. Additional analyses were performed on samples from borings located in potential source areas investigated for the first time during Phase III. Table 2-1 summarizes soil boring samples selected for specific analyses and the CLP laboratory that did the analyses. Results of these analyses are discussed in Section 4 and presented in Appendix E, Table E-1.

2.3 MONITORING WELL INSTALLATION

Monitoring wells were installed in accordance with the procedures in the FSP. From January 5 to January 12, 1993, 10 new monitoring wells were installed in the study area based primarily on LSA sampling results and Phase I and II groundwater results. Typically, the new monitoring wells were installed immediately upgradient, downgradient, and within preliminarily identified source areas to verify these source locations and to provide groundwater quality data necessary for the Risk Assessment (RA) and FS. Locations of Phase III monitoring wells are presented in Figure 2-3.

Monitoring well borings were advanced to various depth intervals by conventional hollow-stem auger drilling techniques. Monitoring well numbers with a "BR" designation are assigned to wells screened on top of bedrock. Soil samples for subsurface stratigraphic description were collected from monitoring well borings using a 24-inch-long, 2-inch or 3-inch OD split-spoon sampler at 5-foot depth intervals, generally beginning at or near the ground surface. The geologist maintained a detailed log of sample depth, blow count, recovery, soil description, and OVA readings. Monitoring well boring logs are presented in Appendix D. Soil samples from screened intervals of the 10 new monitoring well borings were collected and analyzed for grain-size distribution by Aquatec Inc., Colchester, Vermont. Grain-size distribution results are presented in Appendix E. After being described by the

geologist, the split-spoon sample material was containerized with the cuttings for later disposal.

All monitoring wells were constructed with 2-inch ID stainless steel flush-jointed riser pipe and screens. Screens were 10 feet long with 0.01-inch slots of continuous wire-wound design. A filter pack of 100% silica sand was created in the annular space surrounding each screen to approximately 2 feet above the screen. In wells where the top of the screen was located above the water table, a minimum 2-foot bentonite pellet seal was placed above the filter pack and hydrated with potable water. In wells screened below the water table, this sealing procedure was not possible because of clumping and bridging of the pellets. In these wells, a 2-foot seal of the thickest bentonite slurry that could be tremied was placed above the filter pack. Both types of seals were allowed to hydrate until an adequate seal formed. Then bentonite grout was tremied from the seal up to within 2 to 3 feet of the surface as the augers were withdrawn. A 5-foot protective steel cover with a locking cap or flush-mounted well protection was placed over each well and cemented in place to provide security. Table 2-2 presents the specifications of the 10 Phase III monitoring wells. Four shallow, two intermediate, and four deep monitoring wells were installed.

Between samples, the split-spoons were cleaned with a potable water and Liquinox solution using a brush followed by potable water, isopropanol, and distilled water. The drill rig, augers, rods, and ancillary equipment were decontaminated with a steam cleaner between borings, as were all screens and risers before installation. Drill cuttings were contained in 55-gallon drums and subsequently transported to bulk containers for disposal.

Completed wells were allowed to stand for a least 24 hours to allow the grout and concrete pad to set before the wells were developed. The first step of development at each well was to measure the standing water level so that the purge volume could be determined. A Grundfos™ pump was used to purge the water for development. The purged groundwater was tested for pH, conductivity, and temperature, and examined for relative turbidity/clarity. Purging continued until pH, conductivity, temperature, and relative clarity had stabilized, and the well was producing at the maximum achievable rate. In many cases, this required purging from five to 10 well volumes. Water-measurement and pump equipment was decontaminated with a high-pressure steam cleaner between uses. All purge and decontamination water was containerized for later disposal.

2.4 MONITORING WELL SURVEYING AND GROUNDWATER SAMPLING

All monitoring well surveying and groundwater sampling procedures were performed in accordance with the FSP. From January 19 to January 22, 1993, the elevations of the top of the inside casing (TOIC) of the Phase III monitoring wells were surveyed in feet above MSL. The monitoring wells at location MW08 were resurveyed on January 24, 1993, after being converted to flush-mounted wells. Sixty-seven Phase I and II wells were sampled from November 16 to November 19, 1992. From January 26 to January 27, 1993, the 10 Phase III monitoring wells were sampled. Water level measurements were collected prior to each sampling event to document similar groundwater flow patterns during each sampling event.

The TOIC of each Phase III monitoring well was surveyed by E & E with respect to a U.S. Geological Survey benchmark, located on the Ash Road bridge over the St. Joseph River and at an elevation of 735.80 feet above MSL, to a vertical accuracy of ± 0.01 foot (see Table 2-3). The TOIC elevations were used to convert depth to water measurements into elevations above MSL (Table 2-3). The horizontal location of each well was also determined through aerial photographs and a scale map of the study area.

Before sampling, water levels were measured in all wells with a chalked, graduated stainless steel tape; measurements were made both for hydrologic studies and purge volume calculations prior to sampling. Water levels were measured as depth below TOIC. Static water volumes were calculated using the following formula:

$$V = T r^2 (0.163)$$

where:

- V = Static volume of well, in gallons;
- T = Length of water column, in feet;
- r = Inside radius of well casing, in inches; and
- 0.163 = A constant conversion factor.

A maximum of three static water volumes was purged from each well prior to sample collection to ensure that a representative groundwater sample was collected. A KeckSM submersible pump was used to purge all wells. At the end of each purge volume, the water was tested for pH, conductivity, and temperature, and relative turbidity/clarity was noted. After

three volumes were purged and measurements of pH (± 0.25 pH units), conductivity (± 50 $\mu\text{mhos/cm}$), and temperature ($\pm 0.5^\circ$ F) had stabilized, the sample was collected. In the few cases where the wells did not recharge quickly enough to purge three volumes, the wells were purged dry, and the sample was collected when sufficient recharge for an adequate sample volume was obtained.

All groundwater samples, with the exception of the sample collected from monitoring well MW18, were collected with bottom-loading stainless steel bailers. Sample MW18S was collected from a pitcher pump because aboveground damage to the well did not allow a bailer or pump to be placed into the well for sample collection purposes. During collection of all groundwater samples, care was taken to minimize the loss of volatile compounds.

Bailers were decontaminated before each use with a high-pressure steam cleaner, sprayed with isopropanol, rinsed with distilled water, and wrapped in aluminum foil until the next use. The outside of the Keck™ pump and discharge hose were cleaned with the steam cleaner, and the inside was decontaminated with Liquinox™ and distilled water.

Groundwater samples were collected for analysis from each monitoring well. Table 2-4 is a summary of the analysis program for groundwater samples including the analytical fractions submitted for analysis and the CLP laboratory(s) performing the analysis. Results of these analyses are discussed in Section 4 and presented in Appendix E, Table E-2. Samples collected for VOA analysis were preserved with hydrochloric acid and packaged with ice. All sample bottles were labeled, iced, packaged, and shipped in accordance with appropriate procedures stated in the project plans.

Duplicate groundwater samples were collected simultaneously with investigative samples in equal volumes, with the same sampling equipment, and into identical containers. Duplicates were preserved and handled in the same manner as all other groundwater samples. Field blanks were prepared from contaminant-free distilled water that was routed through decontaminated sampling equipment. Field blanks were containerized and handled in the same manner as all other groundwater samples. Duplicate and field blank samples were collected at the rate of one per every 10 or fewer groundwater samples. Trip blanks were prepared by E & E, transported on site by the sampling team, and shipped with the remainder of the samples to the appropriate laboratory at the rate of one per shipping container containing VOA samples per day.

2.5 DRAINAGE NETWORK SAMPLING

On December 18, 1992, five sediment and six water samples were collected from the storm water drainage network on the Conrail facility. Figure 2-4 presents the schematic layout of the drainage network, including drainage flow directions and the sample locations. Sample locations were determined based on locations of known and potentially contributing source areas, the flow path of the network from these source areas, and the ability to locate and access the network via the manholes. All samples were analyzed for TCL VOCs by CLP laboratories. Sediment samples were analyzed by Clayton Laboratory, Novi, Michigan and water samples were analyzed by Southwest Research Institute, San Antonio, Texas. Water samples were preserved with hydrochloric acid and packaged with ice.

Duplicate samples were collected simultaneously with investigative samples in equal volumes, with the same sampling equipment, and into identical containers. Duplicates were preserved and handled in the same manner as all other drainage network samples. Field blanks were prepared from contaminant-free distilled water and handled in the same manner as all other drainage network samples. Duplicate and field blank samples were collected at the rate of one per every 10 or fewer drainage network samples. A trip blank was prepared by E & E, transported to the site by the sampling team, and shipped with drainage network water samples to the appropriate laboratory at the rate of one per shipping container containing VOA samples per day.

To collect the samples, the manhole grates were removed. A bucket auger with extension rods was lowered into the catch basins and sediment was scooped up from the bottom of the drain pipe. The auger was then brought to the surface and sample bottles were filled from the bucket auger with stainless steel spoons. The water samples were collected by taping the sample bottle to the extension rod and lowering it to the bottom of the drain pipe. In some instances, it was necessary to create a depression in the sediment to allow the water to seep into the depression, in order to collect a water sample. A water and a sediment sample were collected at each sampling location with the exception of location DW05, where no sediment was available to sample. Between samples, all equipment was decontaminated with an Alconox™ solution, isopropanol, and distilled water. The results of the analyses are discussed in Section 4 and are presented in Appendix E, Tables E-3 and E-4.

2.6 SURFACE WATER AND SEDIMENT SAMPLING

From December 15 to December 18, 1992, 25 surface water and 25 sediment samples were collected from targeted ecosystems in the study area. The purpose of the sample collection was to identify potential contamination in nearby aquatic and wetland habitats as part of the EA and to obtain sufficient data to evaluate exposure routes as part of the Human Health Evaluation (HHE).

The targeted ecosystems included the St. Joseph River, Baugo Bay, Baugo Creek, and the ponds on the Conrail facility. The sample locations are presented in Figure 2-5. Background surface water and sediment samples were collected from Baugo Creek (SW/SD01 through SW/SD03) and the St. Joseph River (SW/SD17 through SW/SD22) upstream of its intersection with Crawford Ditch. In order to evaluate the ecological impact of contaminant loading via groundwater discharge, surface water and sediment samples were collected from transects within Baugo Bay and the St. Joseph River where they intersect the identified groundwater contamination plume (SW/SD04 through SW/SD16). One surface water sample and one sediment sample were collected from each of the three ponds (SW/SD23 through SW/SD25) on the southern boundary of the Conrail property in order to evaluate the ecological impact of contaminant loading through what appears to be a Conrail facility underground drainage discharge into the ponds. Table 2-5 is a summary of the sampling and analysis program for the surface water and sediment samples including the analytical fractions submitted for analysis and the CLP laboratory(s) performing the analysis. Results of these analyses are discussed in Section 6 and presented in Appendix E, Tables E-5 and E-6.

Surface water and sediment sample collection procedures are in accordance with the FSP. Surface water samples were generally collected with a Kemmerer sampler. For a few near-shore, near-bottom surface water samples, however, sample bottles were immersed in the shallow water for collection. Sediment samples were collected with either a hand-auger or Ekman dredge. All near-bottom, away-from-shore surface water samples and away-from-shore sediment samples were collected between 20 and 100 yards from shore.

Surface water samples collected for VOC analysis were preserved with hydrochloric acid and packed with ice, and surface water samples collected for total metals analysis were preserved with nitric acid. Duplicate surface water and sediment samples were collected simultaneously with investigative samples, in equal volumes, from the same location, with the

same sampling equipment, and into identical containers. Duplicates were preserved and handled in the same manner as all other surface water and sediment samples. Field blanks were prepared from contaminant-free distilled water and handled in the same manner as all other samples. Duplicate and field blank samples were collected at the rate of one per every 10 or fewer samples. Trip blanks were prepared by E & E, transported to the site by the sampling team, and shipped with samples to the appropriate laboratory at the rate of one per shipping container containing VOA samples per day. All sample equipment was decontaminated between samples with an Alconox and water solution; rinsed with potable water, sprayed with isopropanol, and then distilled water; and wrapped for protection when not in use.

2.7 INVESTIGATION-DERIVED WASTE

All drill cuttings from LSA, soil, and monitoring well borings were temporarily stored in 55-gallon drums until the drums could be removed from the drilling locations, usually within one to two days. The drums were then loaded onto a truck by a BC employee using a Bobcat[™] and taken to the E & E field office located at Baugo Creek Park, where they were emptied into a roll-off box. The emptied-out drums were then reused for temporary storage and transfer of investigation-derived waste. The drill cuttings were found to be non-hazardous and the roll-off box contents were properly disposed of. All purge water from LSA groundwater sampling, monitoring well development, monitoring well sampling, and all decontamination water was collected into either a 500-gallon tank or 55-gallon drums and trucked to the E & E field office. The contents of the 500-gallon tank or the drums then were pumped into a 7,000-gallon tanker and tested. The investigative-derived water was properly disposed of at the Elkhart Wastewater Treatment Plant.

Table 2-1					
ANALYSIS PROGRAM FOR PHASE III SOIL BORING SAMPLES					
Boring Number	Depth (feet)	Volatile Organic Analysis	Semivolatile Analysis	Pesticides/ PCBs Analysis	Total Metals Analysis
B40	8 - 10	a	--	--	--
	18 - 20	a	--	--	--
	22 - 24	a	--	--	--
	58 - 60	a	--	--	--
	118 - 120	a	--	--	--
	128 - 130	a	--	--	--
	130 - 132	a	--	--	--
	140 - 142	a	--	--	--
	144 - 146	Da	--	--	--
	148 - 150	a	--	--	--
B41	5 - 7	a	--	--	--
	21 - 23	a	--	--	--
	23 - 25	Da	--	--	--
B42	3 - 5	a	--	--	--
	21 - 23	a	--	--	--
	23 - 25	a	--	--	--
B43	3 - 5	a	--	--	b
	11 - 13	--	a	a	--
	13 - 15	a	a	a	b
	21 - 23	--	Da	Da	b
	23 - 25	a	--	--	--
B44	7 - 9	a	--	--	--
	11 - 13	--	--	--	b
	13 - 15	--	a	a	b
	15 - 17	a	a	a	--

Key at end of table.

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Table 2-1 ANALYSIS PROGRAM FOR PHASE III SOIL BORING SAMPLES					
Boring Number	Depth (feet)	Volatile Organic Analysis	Semivolatile Analysis	Pesticides/ PCBs Analysis	Total Metals Analysis
B44 (Cont.)	17 - 19	—	—	—	b
	19 - 21	—	a	a	—
	23 - 25	a	—	—	—
B45	3 - 5	—	a	a	b
	7 - 9	a	a	a	b
	15 - 17	a	a	a	b
	23 - 25	a	—	—	—
B46	5 - 7	—	a	a	b
	7 - 9	a	a	a	b
	17 - 19	a	a	a	b
	19 - 21	a	—	—	—
B47	0 - 2	Da	—	—	—
	2 - 4	a	—	—	—
	12 - 14	a	—	—	—
B48	4 - 6	Da	—	—	—
	14 - 16	a	—	—	—
	18 - 20	a	—	—	—
B49	6 - 8	Da	—	—	—
	16 - 18	a	—	—	—
	22 - 24	a	—	—	—
B50	0 - 2	a	—	—	—
	4 - 6	a	—	—	—
	12 - 14	a	—	—	—

Key at end of table.

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Table 2-1					
ANALYSIS PROGRAM FOR PHASE III SOIL BORING SAMPLES					
Boring Number	Depth (feet)	Volatile Organic Analysis	Semivolatile Analysis	Pesticides/ PCBs Analysis	Total Metals Analysis
B51	0 - 2	a	--	--	--
	2 - 4	a	--	--	--
	4 - 6	a	--	--	--
B52	0.5 - 1.5	a	--	--	--

Key:

- a Analysis performed by Southwest Research Institute, San Antonio, Texas.
- b Analysis performed by Keystone Laboratory, Houston, Texas.
- D Duplicate sample also submitted for analysis.
- Sample was not submitted for analysis.

Table 2-2			
PHASE III MONITORING WELL SPECIFICATIONS			
Monitoring Well Number	Boring Depth (feet)	Screened Interval (feet BGS)	Date Completed
MW07D	132	121 - 131	1/7/93
MW45	30	18 - 28	1/5/93
MW46S	28	18 - 28	1/5/93
MW46I	65	55 - 65	1/5/93
MW47	40	28 - 38	1/5/93
MW48	30	18 - 28	1/6/93
MW49D	85	75 - 85	1/9/93
MW49BR	144	134 - 144	1/8/93
MW50	121	110 - 120	1/9/93
MW51	113	103 - 113	1/11/93

Note: Well construction details are presented in Appendix D.

Table 2-3

PHASE III GROUNDWATER ELEVATION DATA

Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 15, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW01	Shallow	740.87	14.12	726.75	12.32	728.55	12 - 22
MW02S	Shallow	742.44	14.53	727.91	12.86	729.58	6.5 - 26.5
MW02D	Intermediate	742.30	14.50	727.80	12.78	729.52	70 - 80
MW02BR	Deep	742.53	14.67	727.86	12.97	729.56	158.9 - 168.9
MW03	Shallow	738.80	9.37	729.43	7.33	731.47	5 - 25
MW04S	Shallow	740.80	17.46	723.34	15.79	725.01	15.5 - 35.5
MW04D	Intermediate	741.24	17.96	723.28	16.44	724.80	55 - 65
MW05S	Shallow	734.86	12.46	722.40	11.01	723.85	4.6 - 14.6
MW05D	Intermediate	734.13	11.83	722.30	11.20	722.93	71 - 81
MW06	Shallow	740.18	19.06	721.12	17.91	722.27	10 - 20
MW07	Shallow	731.64	13.16	718.48	12.49	719.15	9 - 19
MW07D	Deep	730.67	N/A	N/A	12.38	718.29	121 - 131
MW08S	Shallow	734.60	15.82	718.78	N/A	N/A	14.5 - 34.5

Key at end of table.

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Table 2-3							
PHASE III GROUNDWATER ELEVATION DATA							
Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 18, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW08D	Intermediate	734.61	15.95	718.66	N/A	N/A	71 - 81
MW08BR	Deep	734.79	16.12	718.67	N/A	N/A	126 - 136
MW08S	Shallow	731.65	N/A	N/A	12.24	719.41	14.5 - 34.5
MW08D	Intermediate	731.57	N/A	N/A	12.28	719.29	71 - 81
MW08BR	Deep	731.93	N/A	N/A	12.65	719.28	126 - 136
MW09	Shallow	740.43	20.73	719.70	19.86	720.57	15 - 25
MW10S	Shallow	728.70	11.15	717.55	10.84	717.86	9 - 19
MW10D	Intermediate	728.26	7.93	720.33	6.90	721.36	71 - 81
MW11S	Shallow	739.50	Dry	N/A	15.94	723.56	10 - 20
MW11D	Intermediate	739.28	16.64	722.64	15.78	723.50	54.5 - 64.5
MW12	Shallow	741.59	11.54	730.05	9.79	731.80	10 - 20
MW13S	Shallow	750.20	12.50	737.70	10.94	739.26	10 - 30
MW13D	Intermediate	750.50	12.75	737.75	11.24	739.26	70 - 80

Key at end of table.

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Table 2-3

PHASE III GROUNDWATER ELEVATION DATA

Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 15, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW14	Shallow	740.87	6.50	734.37	4.93	735.94	4 - 24
MW15	Deep	742.44	5.47	736.97	4.11	738.33	90 - 100
MW16	Shallow	743.51	4.93	738.58	4.27	739.24	4.5 - 24.5
MW18	Shallow	747.18	5.40	741.78	3.81	743.37	9.5 - 19.5
MW19S	Shallow	752.30	13.28	739.02	11.97	740.33	10 - 30
MW19D	Intermediate	752.37	13.28	739.09	12.04	740.33	60 - 70
MW20S	Shallow	748.41	12.67	735.74	11.59	736.82	11 - 31
MW20D	Intermediate	748.65	12.56	736.09	11.44	737.21	71 - 81
MW21S	Shallow	754.83	14.03	740.80	13.02	741.81	14.5 - 34.5
MW21D	Intermediate	754.87	14.16	740.71	13.14	741.73	70 - 80
MW23S	Shallow	741.82	8.58	733.24	6.92	734.90	10 - 30
MW23D	Intermediate	742.29	9.97	732.32	8.32	733.97	70 - 80
MW24	Shallow	745.12	10.68	734.44	9.08	736.04	10 - 30

Key at end of table.

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Table 2-3

PHASE III GROUNDWATER ELEVATION DATA

Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 15, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW25	Shallow	743.72	9.84	733.88	8.21	735.51	10 - 20
MW26	Shallow	752.02	15.58	736.44	14.24	737.78	10 - 30
MW27S	Shallow	751.87	11.58	740.29	10.30	741.57	8 - 18
MW27I	Intermediate	752.13	12.22	739.91	10.95	741.18	43.1 - 53.1
MW28S	Shallow	750.83	12.26	738.57	10.74	740.09	8.5 - 18.5
MW28I	Intermediate	750.91	12.50	738.41	11.11	739.80	43 - 53
MW29S	Shallow	751.77	11.17	740.60	10.09	741.68	8 - 18
MW29I	Intermediate	752.37	11.79	740.58	10.63	741.74	35.3 - 45.3
MW30S	Shallow	748.13	9.46	738.67	8.06	740.07	8 - 18
MW30I	Intermediate	748.18	10.10	738.08	8.77	739.41	42.7 - 52.7
MW30D	Deep	748.09	10.04	738.05	8.67	739.42	94.7 - 104.7
MW30BR	Deep	747.94	9.89	738.05	8.54	739.40	137 - 147
MW31S	Shallow	751.45	11.12	740.33	9.96	741.49	8.5 - 18.5

Key at end of table.

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Table 2-3

PHASE III GROUNDWATER ELEVATION DATA

Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 15, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW311	Intermediate	751.82	11.51	740.31	10.35	741.47	42 - 52
MW32S	Shallow	746.99	7.15	739.84	5.97	741.02	18 - 28
MW32I	Intermediate	746.93	7.20	739.73	6.00	740.93	40 - 50
MW33S	Shallow	745.40	7.79	737.61	6.30	739.10	16.5 - 26.5
MW33I	Intermediate	745.31	7.00	738.31	6.53	738.78	35 - 45
MW34	Intermediate	744.33	9.58	734.75	8.04	736.29	40 - 50
MW35	Shallow	748.50	7.68	740.82	6.61	741.89	20 - 30
MW36I	Intermediate	747.04	8.36	738.68	7.08	739.96	45 - 55
MW37S	Shallow	741.47	16.66	724.81	15.09	726.38	12 - 22
MW37D	Deep	741.36	16.52	724.84	14.97	726.39	90 - 100
MW38S	Shallow	737.15	15.74	721.41	14.43	722.72	11 - 21
MW38D	Deep	736.84	15.43	721.41	14.13	722.71	90 - 100
MW39	Shallow	752.88	13.78	739.10	12.58	740.30	20 - 30

Key at end of table.

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Table 2-3

PHASE III GROUNDWATER ELEVATION DATA

Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 18, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW40	Shallow	753.40	14.40	739.00	13.26	740.14	20 - 30
MW41	Intermediate	741.55	7.86	733.69	6.29	735.26	65 - 75
MW42I	Intermediate	742.19	10.87	731.32	9.17	733.02	40.1 - 50.1
MW43S	Shallow	728.92	8.09	720.83	6.75	722.17	6 - 16
MW43BR	Deep	728.60	9.28	719.32	8.42	720.18	146.5 - 156.5
MW44D	Deep	739.71	17.10	722.61	15.62	724.09	115 - 125
MW45	Shallow	760.67	N/A	N/A	18.90	741.77	18 - 28
MW46S	Shallow	747.05	N/A	N/A	6.07	740.98	18 - 28
MW46I	Intermediate	747.24	N/A	N/A	6.27	740.97	55 - 65
MW47	Shallow	745.55	N/A	N/A	6.67	738.88	28 - 38
MW48	Shallow	751.12	N/A	N/A	9.57	741.55	18 - 28
MW49D	Intermediate	745.62	N/A	N/A	6.31	739.31	75 - 85
MW49BR	Deep	745.61	N/A	N/A	6.28	739.33	134 - 144

Key at end of table.

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Table 2-3

PHASE III GROUNDWATER ELEVATION DATA

Monitoring Well Number	Zone Interval for Potentiometric Contour Maps	Top of Inner Casing Elevation (feet above Mean Sea Level)	November 15, 1992		January 23-24, 1993		Screened Interval (feet BGS)
			Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	Depth to Water (feet)	Water Level Elevation (feet above Mean Sea Level)	
MW50	Deep	737.42	N/A	N/A	14.71	722.71	110 - 120
MW51	Deep	741.09	N/A	N/A	4.49	736.60	103 - 113

Key:

N/A Not applicable.

Note: The St. Joseph River elevation (in feet above MSL) at the Ash Road bridge was 715.61 on November 15, 1992, and 715.65 on January 23-24, 1993.

Table 2-4 ANALYSIS PROGRAM FOR PHASE III GROUNDWATER SAMPLES		
Monitoring Well Number	Date Sampled	Volatile Organic Analysis
MW01	11/16/92	a
MW02S	11/17/92	a
MW02D	11/17/92	a
MW02BR	11/17/92	a
MW03	11/16/92	a
MW04S	11/16/92	a
MW04D	11/16/92	a
MW05S	11/16/92	a
MW05D	11/16/92	a
MW06	11/16/92	a
MW07S	11/16/92	a
MW07D	1/27/93	b
MW08S	11/17/92	a
MW08D	11/17/92	a
MW08BR	11/17/92	a
MW09	11/16/92	a
MW10S	11/16/92	a
MW10D	11/16/92	a
MW11S	11/17/92	a
MW11D	11/17/92	a
MW12	11/17/92	a
MW13S	11/17/92	Da
MW13D	11/17/92	a
MW14	11/19/92	a
MW15	11/16/92	a

Key at end of table.

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<p>Table 2-4</p> <p>ANALYSIS PROGRAM FOR PHASE III GROUNDWATER SAMPLES</p>		
Monitoring Well Number	Date Sampled	Volatile Organic Analysis
MW16	11/16/92	a
MW18	11/19/92	a
MW19S	11/16/92	a
MW19D	11/16/92	a
MW20S	11/17/92	a
MW20D	11/17/92	a
MW21S	11/16/92	Da
MW21D	11/16/92	a
MW23S	11/17/92	a
MW23D	11/17/92	a
MW24	11/18/92	a
MW25	11/18/92	a
MW26	11/18/92	a
MW27S	11/18/92	a
MW27I	11/18/92	a
MW28S	11/18/92	a
MW28I	11/18/92	a
MW29S	11/18/92	a
MW29I	11/18/92	a
MW30S	11/18/92	a
MW30I	11/18/92	Da
MW30D	11/18/92	a
MW30BR	11/18/92	a
MW31S	11/18/92	a
MW31I	11/18/92	a
MW32S	11/18/92	a

Key at end of table.

05-ZF3901_CX388-03/21/94-01

Table 2-4 ANALYSIS PROGRAM FOR PHASE III GROUNDWATER SAMPLES		
Monitoring Well Number	Date Sampled	Volatile Organic Analysis
MW32I	11/18/92	a
MW33S	11/18/92	a
MW33I	11/18/92	a
MW34	11/18/92	Da
MW35	11/18/92	a
MW36	11/18/92	a
MW37S	11/19/92	a
MW37D	11/19/92	a
MW38S	11/19/92	a
MW38D	11/19/92	Da
MW39	11/17/92	a
MW40	11/17/92	a
MW41	11/19/92	Da
MW42	11/19/92	a
MW43S	11/17/92	a
MW43DR	11/17/92	Da
MW44	11/17/92	a
MW45	1/26/93	b
MW46S	1/26/93	Db
MW46I	1/26/93	b
MW47	1/26/93	b
MW48	1/26/93	b
MW49D	1/26/93	b
MW49DR	1/26/93	b
MW50	1/27/93	b

Key at end of table.

0056

Table 2-4 ANALYSIS PROGRAM FOR PHASE III GROUNDWATER SAMPLES		
Monitoring Well Number	Date Sampled	Volatile Organic Analysis
MW51	1/26/93	Db

Key:

- a Analysis performed by Natex/Pacific Northwest Environmental Lab, Redmond, Washington.
- b Analysis performed by University of Iowa Hygienic Laboratory, Iowa City, Iowa.
- D Duplicate sample also submitted for analysis.

Table 2-5 ANALYSIS PROGRAM FOR SURFACE WATER AND SEDIMENT SAMPLES				
Sample Number	Volatile Organic Analysis	Semivolatile Analysis	Pesticides/PCB Analysis	Total Metals Analysis
SW01	a	a	a	b
SW02	a	a	a	b
SW03	a	a	a	b
SW04	a	a	a	b
SW05	a	a	a	b
SW06	a	a	a	b
SW07	a	a	a	b
SW08	a	a	a	b
SW09	Da	Da	Da	D6
SW10	a	a	a	b
SW11	a	a	a	b
SW12	a	a	a	b
SW13	Da	Da	Da	D6
SW14	a	a	a	b
SW15	a	a	a	b
SW16	a	a	-	b
SW17	a	a	a	b
SW18	a	a	a	b
SW19	a	a	a	b
SW20	a	a	a	b
SW21	a	a	a	b
SW22	a	a	a	b
SW23	a	a	a	b
SW24	a	a	a	b
SW25	Da	Da	Da	D6
SD01	c	c	c	d

Key at end of table.

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Table 2-5				
ANALYSIS PROGRAM FOR SURFACE WATER AND SEDIMENT SAMPLES				
Sample Number	Volatile Organic Analysis	Semivolatile Analysis	Pesticides/PCB Analysis	Total Metals Analysis
SD02	c	c	c	d
SD03	c	c	c	d
SD04	c	c	c	d
SD05	c	c	c	d
SD06	c	c	c	d
SD07	c	c	c	d
SD08	c	c	c	d
SD09	Dc	Dc	Dc	Dd
SD10	c	c	c	d
SD11	c	c	c	d
SD12	c	c	c	d
SD13	Dc	Dc	Dc	Dd
SD14	c	c	c	d
SD15	c	c	c	d
SD16	c	c	c	d
SD17	c	c	c	d
SD18	c	c	c	d
SD19	c	c	c	d
SD20	c	c	c	d
SD21	c	c	c	d
SD22	c	c	c	d
SD23	c	c	c	d
SD24	c	c	c	d

Key at end of table.

05-ZF7901_C0388-03/21/94-D1

ANALYSIS PROGRAM FOR SURFACE WATER AND SEDIMENT SAMPLES

Sample Number	Volatile Organic Analysis	Semivolatile Analysis	Pesticides/PCB Analysis	Total Metals Analysis
SD25	Dc	Dc	Dc	Dd

a Analysis performed by Southwest Research Institute, San Antonio, Texas.
b Analysis performed by Dataschem Laboratories, Inc., Salt Lake City, Utah.
c Analysis performed by Clayton Environmental Consultants, Novi, Michigan.
d Analysis performed by Keystone Lab-Houston, Houston, Texas.
D Duplicate sample also submitted for analysis.
- Analysis was not performed.

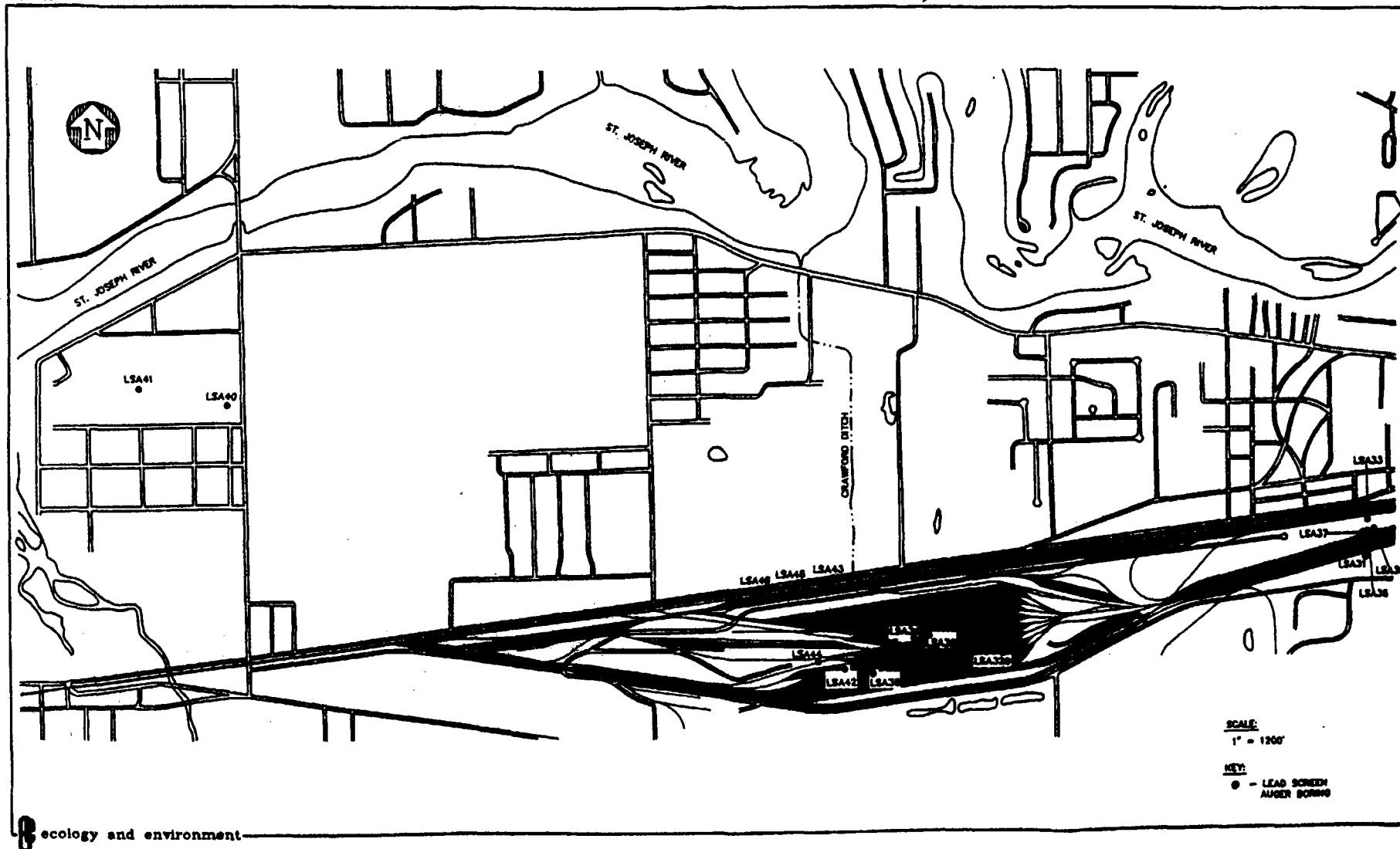
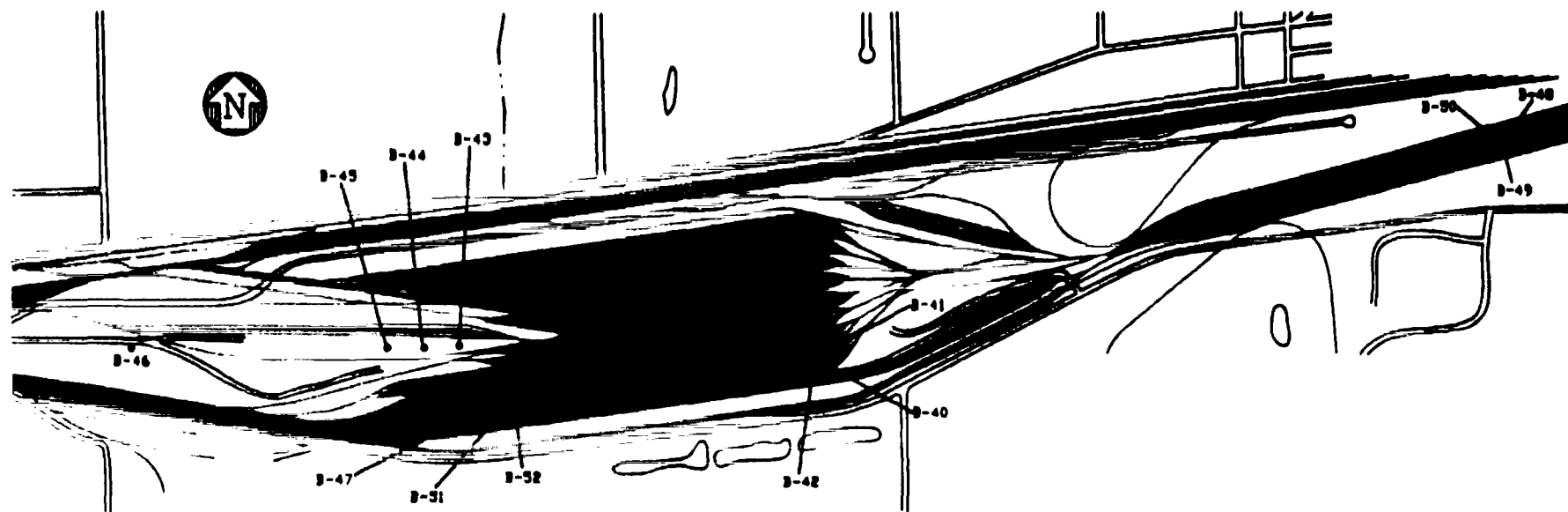


FIGURE 2-1

PHASE III
LEAD-SCREEN
AUGER BORING
LOCATION MAP

2-29



KEY
• - SOIL BORING LOCATION

SCALE
1" = 70'

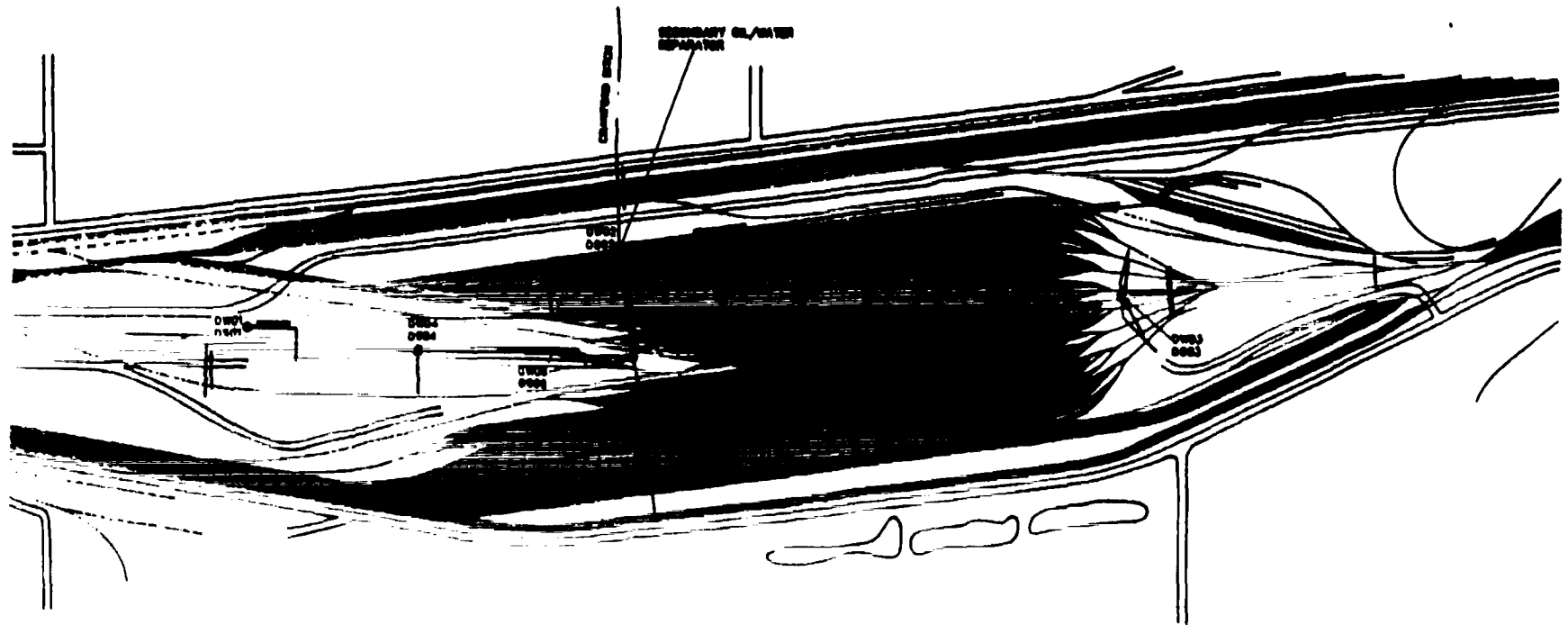
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FIGURE 2-2

PHASE II SOIL BORING LOCATIONS MAP

2-20





SCALE
1" = 300'

LEGEND:

- - SAMPLE LOCATION
- - DIRECTION OF FLOW
- OW - DRAINAGE WATER
- OS - DRAINAGE SEGMENT

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Source: Adapted from Canal construction diagram of drainage network.

FIGURE 2-4

DRAINAGE NETWORK
SAMPLE LOCATIONS
MAP

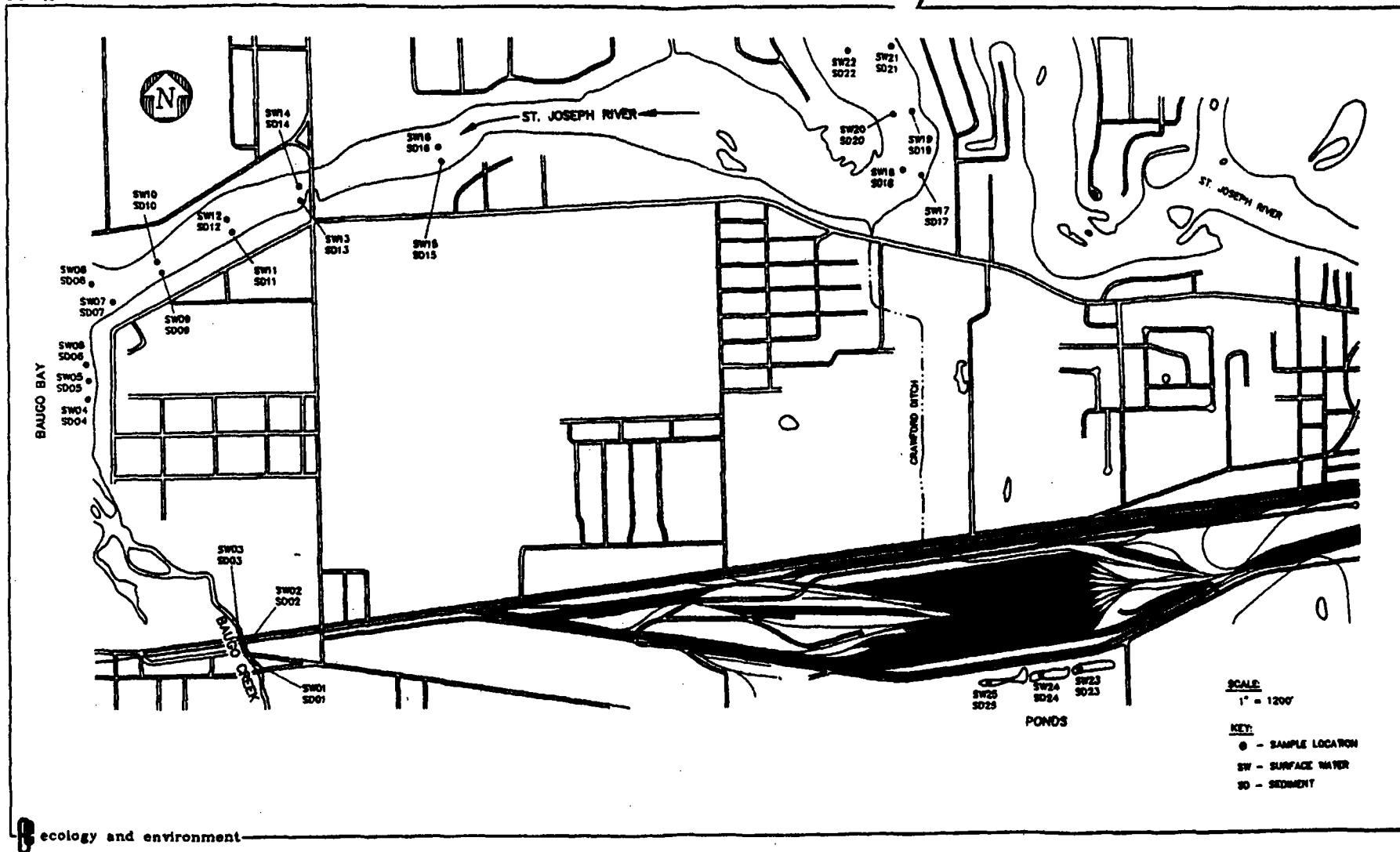


FIGURE 2-5 SURFACE WATER AND
SEDIMENT SAMPLE
LOCATIONS

3. PHYSICAL CHARACTERISTICS OF THE SITE

This section summarizes geotechnical information and data collected by E & E during the Phase III RI. This section also synthesizes data from all three phases of the RI and presents a site-specific geologic and hydrogeologic analysis. Information compiled from the examination of aquifer materials collected during drilling activities has been used to describe the stratigraphy in the study area. Laboratory results of grain-size distributions for selected samples of the aquifer materials have been used to further characterize the geologic and hydrogeologic conditions in the study area. Static water level measurements and slug test results have been used to calculate hydrogeologic parameters.

3.1 GEOLOGY

The information collected by E & E during the subsurface investigations is used to describe geological conditions present in the study area. The 52 soil borings and 77 boreholes for monitoring well installation allowed for extensive coverage, with respect to area and depth, of the study area. The combined results of the subsurface soil investigations that were conducted during the three phases of the RI show that the study area consists of unstratified sand and gravel glacial outwash deposits (see Appendices B, D, and F for drilling logs). Figure 3-1 illustrates the locations of two cross-sections of the site; the cross-sections are presented as Figures 3-2 and 3-3. Drilling logs were used to construct these cross-sections located within the study area. The cross-sections also show the screened interval of each well or well nest.

Figure 3-2 is cross-section A-A' with an approximate east-west trend that parallels U.S. 33. This cross-section extends from MW12 to MW40, and was constructed to illustrate the extent of a silt and clay unit identified in the study area that is centered near MW15.

Although this silt and clay has variable color and clay content, in general, it can be correlated from MW15 eastward to MW18, westward to MW51, and southeastward to MW36 (see Figure 3-3).

Figure 3-3 is cross-section B-B' along a line bearing approximately southwest to northeast, from monitoring well location MW13 to MW16. This geologic cross-section was constructed because it is approximately perpendicular to the groundwater flow direction. Also refer to Figure 3-1 of the *Phase II Technical Memorandum* (Appendix B), which is a cross-section that extends across the railyard to the St. Joseph River, and is oriented parallel to the groundwater flow direction. Evaluation of the lateral continuity of clay and silt units shows that the silt and clay is present as discrete lenses or masses. Figure 3-2, Figure 3-3, Figure 3-1 of the *Phase II Technical Memorandum* (Appendix B), and the drilling logs clearly show that there is no continuous confining layer present in the study area. Below the discontinuous silts and clays, the study area is dominated by interbedded brown sand and brown sand and gravel.

Table 3-1 shows the results of laboratory grain-size analyses for soil samples collected during the installation of the Phase III monitoring wells. Ten samples were submitted for grain-size analysis; Table 3-1 summarizes the results, which are presented in Appendix E, and applies the Unified Soil Classification System (ASTM D 2487-85) to the laboratory data. Table 3-1 also lists the results obtained for specific gravity because this factor is utilized in mass transport calculations. The specific gravity ranges from 2.61 to 2.72 with the average being 2.67. A value of 2.65 is normal for most sandy soils (Domenico and Schwartz 1990). The soil samples were collected from the depth of the screened intervals of the Phase III monitoring wells. The samples consist of soil that was collected from intervals which range from 1 foot to 6 feet. Because the length of all Phase III monitoring well screens is 10 feet, the samples do not reflect the entire interval over which the monitoring wells are screened. Although it is rare for the grain-size distribution of the aquifer material to be constant over a 10-foot depth interval, these samples provide a reliable characterization of the soil type in which the Phase III monitoring wells are screened.

The predominant soil type shown in Table 3-1 is poorly graded sand (SP), based upon the terminology used by the Unified Soil Classification System. For practical purposes, the sand and sand and gravel unit (in which most of the monitoring wells from all three phases

3.2 HYDROGEOLOGY

The depth to the water table in the study area varies from approximately 3 feet BGS to nearly 20 feet BGS. The observed depth to water depends on geographic location, season, and elevation of the ground surface. The Phase III water level elevation data collected from all monitoring wells were presented in Table 2-3. From December 1989 to January 1993, water level measurements were collected from the existing wells on at least 14 separate occasions. A comparison of the data recorded over this three-year time span indicates fluctuations of less than 3 feet observed in the elevation of the potentiometric surface. The relative static water levels among wells were consistent for each monitoring event, causing the shape of the potentiometric contour lines to remain constant.

Groundwater table surface maps and intermediate and deep potentiometric surface maps were constructed to interpret groundwater flow patterns in the unconfined aquifer using November 15, 1992 (see Figures 3-4 through 3-6) and January 23 and 24, 1993 (see Figures 3-7 through 3-9) data. The maps were constructed using the data from Table 2-3 to enable comparison of the three zone depths in the unconfined aquifer. The shallow zone extends from the water table to approximately 35 feet BGS. The intermediate zone spans 35 feet BGS to 85 feet BGS. The deep zone extends from 85 feet BGS to the top of bedrock. Table 2-3 indicates the monitoring zone of each well. Water level data from monitoring wells screened within these respective zones were used to construct the respective maps. Comparison of the November 15, 1992, set of maps (Figures 3-4 through 3-6) with the January 23 and 24, 1993, set (Figures 3-7 through 3-9) shows similar flow patterns for comparable zones. Both sets of maps are also consistent with the shape of potentiometric contour lines determined during Phases I and II. The median Phase III horizontal groundwater gradient is 0.0020 ft/ft for the shallow zone, 0.0019 ft/ft for the intermediate zone, and 0.0020 ft/ft for the deep zone.

Figures 3-4 and 3-7 show contour lines that represent the groundwater table in November 1992 and January 1993, respectively. Groundwater mounds are present at monitoring well locations MW18, MW30S, MW43S, and MW45. The MW18, MW30S, and MW45 mounds may be related to the clay unit beneath the northern portion of the classification yard, as shown in Figures 3-2 and 3-3. This relationship between clay and mounding is not consistent throughout the study area because other areas where clay or silt is present do not exhibit mounding. An additional explanation for the groundwater mounding near MW18

and MW45 is based on the potential for greater recharge due to the adjacent open field near these wells. The proximity and shape of the St. Joseph River is also a contributing factor to the curvature in the contours that is observed in relation to monitoring well location MW18. MW45 is located within the railyard property and is at a ground surface elevation that is approximately 8 feet higher than nearby wells located along U.S. 33 or south of the railroad tracks. Two factors may be responsible for the groundwater mound at MW43S. Clayey silt extends from 14 feet to 18 feet BGS at this location and the bottom of the screened interval for MW43S is 16 feet BGS. The other factor is a pond that is located next to this well that may be acting as a localized recharge source to groundwater. The general groundwater flow direction is to the west-northwest. In the LaRue Street area, however, the general flow direction is north.

Figures 3-5 and 3-6 show contour lines that represent the potentiometric surface of the intermediate and deep monitoring zones for November 1992. Figures 3-8 and 3-9 are additional maps constructed using data from January 1993. These four maps are similar to the two corresponding groundwater table maps with respect to the spacing of the contour lines. For the geologic conditions in the Conrail Site study area, this indicates that the three monitoring zones exhibit similar hydraulic conductivities. Comparison of the shape of the contour lines for the shallow and intermediate zones reveals general similarity with the exception that the intermediate maps cannot be used to evaluate the presence of the groundwater mounds at monitoring well locations MW18, MW43S, and MW45 shown using the water table maps. This is due to the fewer locations of wells within the intermediate monitoring zone. The groundwater mound located at monitoring well MW30S is not present in the intermediate and deep zones. The mound present in the shallow zone at MW43S is not observed in the deep zone. Assuming the groundwater flow direction is at right angles to the contour lines, the similarity in shape of the contour lines among the three zones indicates similar groundwater flow paths.

Table 3-2 lists the vertical hydraulic gradients for the two Phase III water elevation data sets. The vertical hydraulic gradient (I) between two wells at a nested location was calculated using the following equation:

$$I = \frac{H_a - H_b}{I_b - I_a} \quad [\text{Eq. 3-1}]$$

where H_a is the water level elevation in the shallower of the two wells, and H_b is the water level elevation in the deeper well. The I_b and I_a terms represent the midpoints (depth BGS) of the screened intervals of the deeper and shallower wells, respectively.

These results show the same general downward gradient (as evidenced by the "+" signs) in the study area that has been observed during the previous phases. The vertical hydraulic gradients and the respective locations of the monitoring well nests in the study area are consistent with groundwater recharge in the railyard and subsequent groundwater discharge to the St. Joseph River.

Hydraulic conductivity values were calculated from slug test data collected during the Phase II investigation, and correspond to the filter pack and aquifer material immediately surrounding the screened interval of the tested well. These values may be influenced by disturbances caused by monitoring well installation, such as intrusion of drilling mud into the formation. It was noted in the *Phase II Technical Memorandum* that the wells installed using the mud rotary technique displayed the lowest values of hydraulic conductivity. It is not known whether the lower hydraulic conductivity values measured in the mud rotary-installed wells are a reflection of the use of this technique or a difference in the geology at depth. A causal relationship between mud rotary wells and low slug test results cannot be deduced because the wells installed to monitor the deepest zone of the aquifer were the same wells installed using the mud rotary technique. Because the distribution, range, and magnitude of the slug test results are typical values for unconsolidated sand aquifers (EPA 1990b), the horizontal groundwater velocity will be estimated in a manner that allows for the results to be applicable to all zone depths.

As a basis for comparison, the hydraulic conductivity value derived from the pump test conducted in the study area by a water supply contractor (Peerless-Midwest, Inc. [no date]), can be utilized. A hydraulic conductivity value calculated from a pump test represents the hydraulic conductivity of the aquifer material within the zone of influence of the pumping. Hence, the impact of the disturbances associated with the monitoring well installation, as described for the slug test, is reduced proportionally to the radius of influence of the

pumping. For these reasons and because of the heterogeneity of the aquifer, variation between slug test data and a large-scale pump test's data within one or two orders of magnitude is not unusual.

Horizontal velocity (V_h) of groundwater may be calculated using the following equation:

$$V_h = \frac{K_h \times i_h}{n_e} \quad [\text{Eq. 3-2}]$$

where K_h is the horizontal hydraulic conductivity, i_h is the horizontal gradient, and n_e is the effective porosity value (Domenico and Schwartz 1990). For the horizontal hydraulic conductivity values in the above equation, E & E's Phase II slug test results and pump test data from Peerless-Midwest, Inc., can be used for comparative purposes. Table 3-3 presents selected calculated horizontal velocity values.

Table 3-3 was developed to illustrate minimum, mean, and maximum estimated travel times for groundwater based upon the ranges observed in the data sets for the input parameters. The hydraulic conductivity values range from the lowest to highest observed Phase II slug test results. A value that is within this range is the pump test result from Peerless-Midwest, Inc. (no date), which serves as a comparison to the slug test method of measurement. The geometric mean of the Phase II slug test results is also shown. The range and median of the horizontal gradient were derived from Figures 3-4 through 3-9. The range in effective porosity was estimated by comparing the aquifer material descriptions derived from grain-size analysis in Table 3-1 with representative values listed by Imbrigiotta and Martin (1981) and EPA (1983). The lower limit of velocity was calculated using the lowest hydraulic conductivity, lowest horizontal gradient, and the largest effective porosity values, whereas the upper limit was calculated using the highest hydraulic conductivity, highest gradient, and lowest effective porosity. The large range in groundwater travel times, as shown in Table 3-3, is common for this calculation, and is due to the variation in the estimated hydraulic conductivity, horizontal gradient, and effective porosity values of the aquifer. The variation in the hydraulic conductivity value is due to heterogeneity of the glacial material. It is important to note that Table 3-3 shows velocities and travel times for

groundwater, not contaminants. The effect of retardation caused by the adsorption of volatile organic compounds onto organic carbon has not been taken into account in the travel times shown in Table 3-3. This concept will be presented and discussed in Section 5. The travel times were calculated to establish a baseline for estimating the time required for groundwater to migrate from a potential source to an arbitrary downgradient location and to estimate the time required for groundwater to travel a given distance. Values from Table 3-3 will be utilized in Section 5 in calculations supporting the potential that sources of the groundwater contamination in the study area exist on the Conrail facility.

Table 3-1

**GRAIN-SIZE DISTRIBUTION RESULTS FOR SOIL SAMPLES COLLECTED
DURING THE INSTALLATION OF THE PHASE III MONITORING WELLS**

Monitoring Well Number	Depth of Soil Sample (feet)	Specific Gravity	Percent Gravel	Percent Sand	Percent Fines	Unified Soil Classification System Group Symbol	Unified Soil Classification System Group Name
MW07D	124 - 126	2.64	15.8	80.3	3.9	SP	Poorly graded sand with gravel
MW45	22.5 - 24.5	2.62	4.6	90.2	5.2	SP-SM	Poorly graded sand with silt
MW46S	21 - 27	2.72	13.2	59.0	27.8	SM	Silty sand
MW46I	56 - 62	2.65	0.0	87.9	12.1	SP/SW-SM	Poorly to well-graded sand with silt
MW47	32.5 - 34.5	2.66	34.5	59.7	5.8	SW-SM	Well-graded sand with gravel and silt
MW48	21 - 22	2.61	0.0	97.0	3.0	SP	Poorly graded sand
MW49D	80 - 82	2.71	10.3	83.4	6.3	SP/SW-SM	Poorly to well-graded sand with silt
MW49BR	138 - 140	2.67	8.6	89.1	2.3	SP	Poorly graded sand
MW50	114 - 116	2.69	23.8	73.3	2.9	SP/SW	Poorly to well-graded sand with gravel
MW51	106 - 108	2.67	15.2	82.1	2.7	SP	Poorly graded sand with gravel

Table 3-2		
PHASE III VERTICAL HYDRAULIC GRADIENTS		
Well Nos.	November 15, 1992	January 23-24, 1993
MW02S-MW02D	+.00188	+.00103
MW02S-MW02BR	+.00034	+.00014
MW02D-MW02BR	-.00067	-.00045
MW04S-MW04D	+.00174	+.00609
MW05S-MW05D	+.00151	+.01386
MW07S-MW07D	N/A	+.00768
MW08S-MW08D	+.00233	+.00233
MW08S-MW08BR	+.00103	+.00122
MW08D-MW08BR	-.00018	+.00018
MW10S-MW10D	-.04484	-.05645
MW11S-MW11D	N/A	+.00135
MW13S-MW13D	-.00091	+.00000
MW19S-MW19D	-.00156	+.00000
MW20S-MW20D	-.00636	-.00709
MW21S-MW21D	+.00178	+.00158
MW23S-MW23D	+.01673	+.01691
MW27S-MW27I	+.01083	+.01111
MW28S-MW28I	+.00464	+.00841
MW29S-MW29I	+.00073	+.00220
MW30S-MW30I	+.01700	+.01902
MW30S-MW30D	+.00715	+.00750
MW30S-MW30BR	+.00481	+.00519
MW30I-MW30D	+.00058	-.00019
MW30I-MW30BR	+.00032	+.00011
MW30D-MW30BR	+.00000	+.00047
MW31S-MW31I	+.00060	+.00060

Key at end of table.

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Table 3-2 PHASE III VERTICAL HYDRAULIC GRADIENTS		
Well Nos.	November 15, 1992	January 23-24, 1993
MW32S-MW32I	+ .00500	+00409
MW33S-MW33I	-.03784	+ .01730
MW37S-MW37D	-.00038	-.00013
MW38S-MW38D	+ .00000	+ .00013
MW43S-MW43BR	+ .01075	+ .01416
MW46S-MW46I	N/A	+ .00027
MW49D-MW49BR	N/A	-.00034

Key:

N/A Data not available.

Table 3-3

ESTIMATION OF THE HORIZONTAL FLOW VELOCITY AND TRAVEL TIME FOR GROUNDWATER

Hydraulic Conductivity			Horizontal Gradients (ft/ft) derived from Figures 3-4 Through 3-9	Effective Porosity ^a	Velocity (feet/year)	Travel Time (Track 69 to Intersection of C.R. 1 and U.S. 33 ~4000 ft.)	Travel Time (from Intersection of C.R. 1 and U.S. 33 to the St. Joseph River ~7750 ft.)
Data Source	(cm/s)	(ft/day)					
Lowest E & E Phase II Slug Test Result	2.9×10^{-3}	8.3	0.0013	0.35	11	360 years	690 years
Geometric mean of E & E Phase II Slug Test Results	2.4×10^{-2}	69	0.0020	0.25	200	20 years	38 years
Peerless-Midwest Pump Test Result	9.8×10^{-2}	280	0.0020	0.25	810	4.9 years	9.5 years
Highest E & E Phase II Slug Test Result	1.1×10^{-1}	300	0.0030	0.15	2,200	1.8 years	3.5 years

^a Estimates based on grain-size analysis data and representative literature values with ranges chosen to illustrate expected variation.

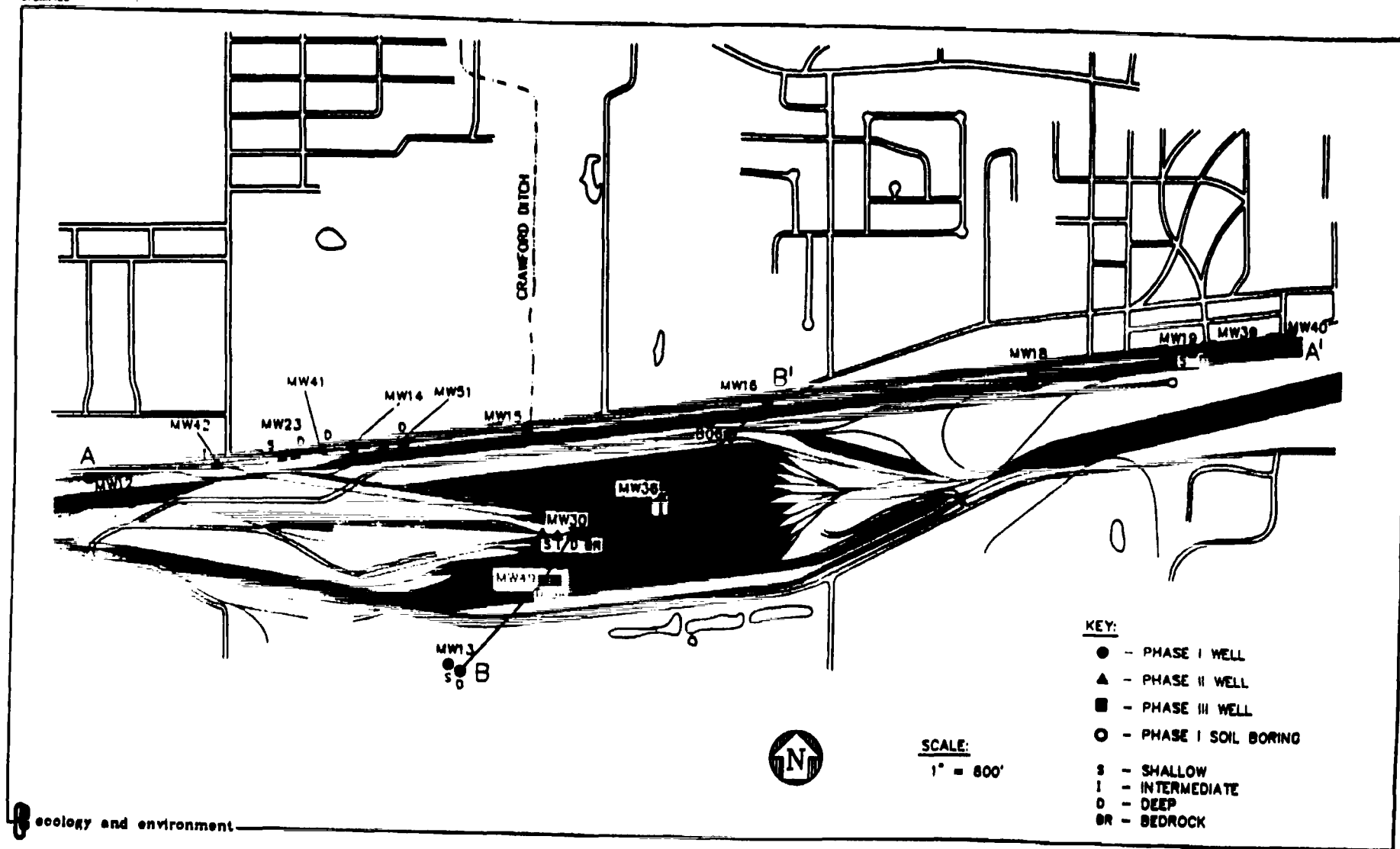
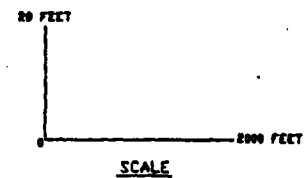
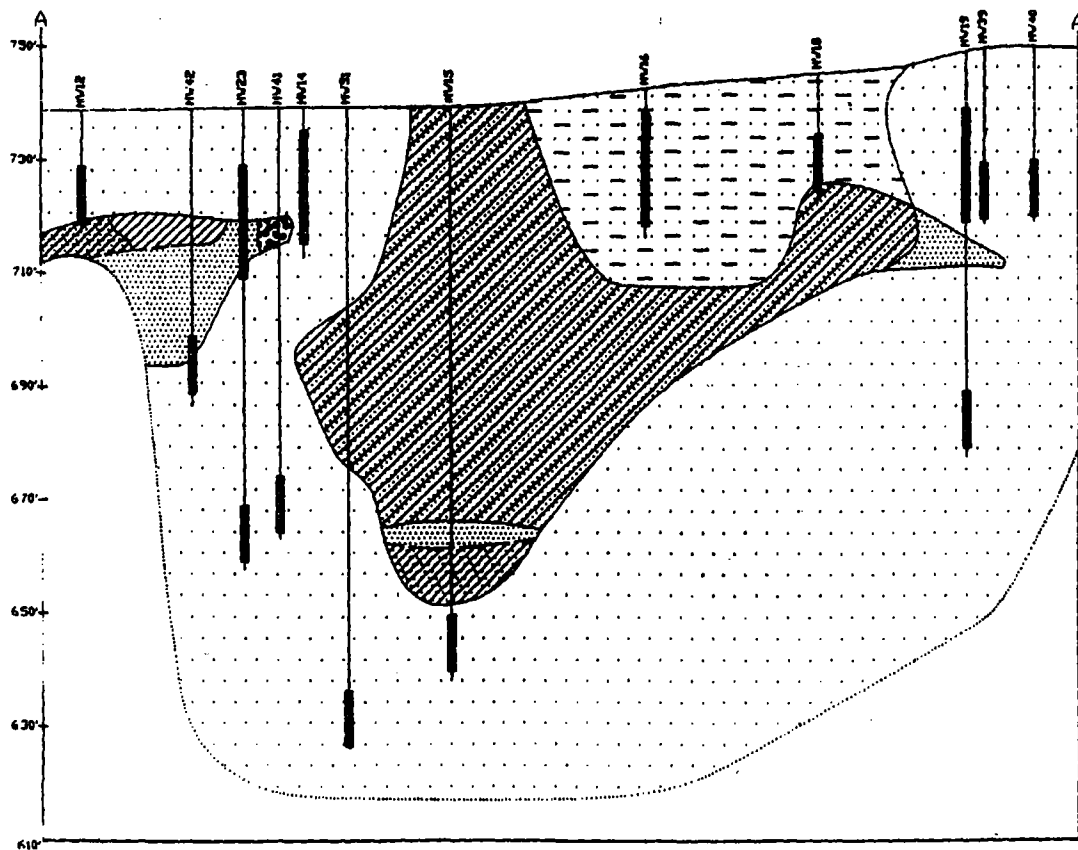








FIGURE 3-1
CROSS-SECTION LOCATIONS -
A-A' and B-B'
MAP

3-13



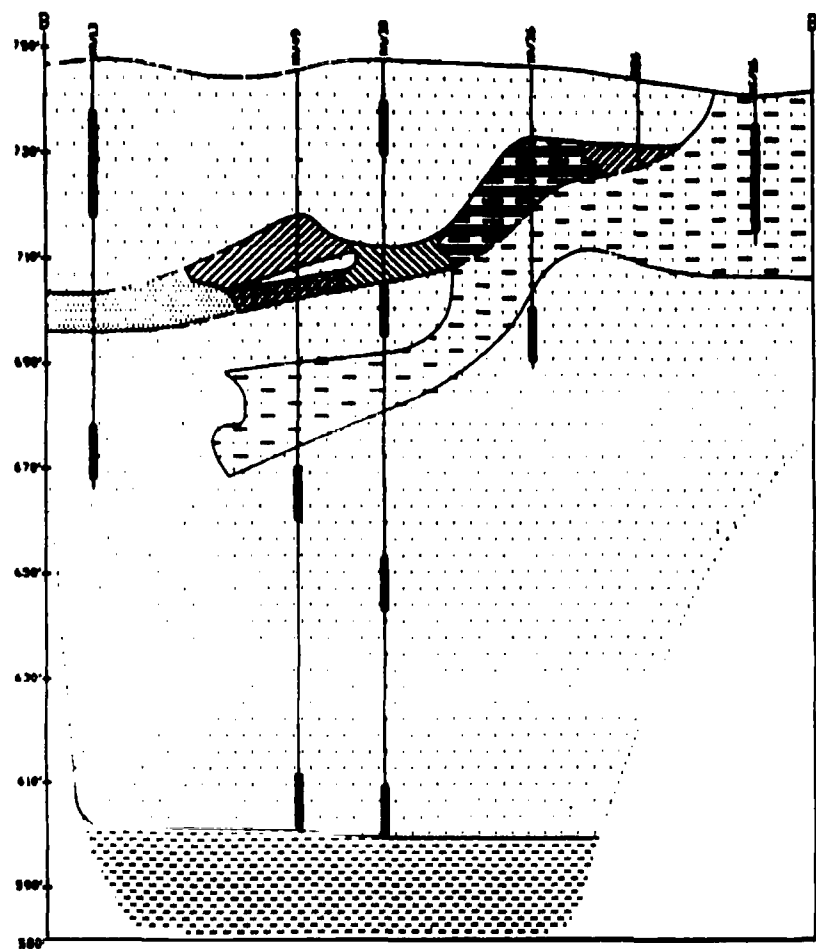
LEGEND

-  - BROWN GRAVEL
-  - BROWN SAND WITH LOCALIZED SAND AND GRAVEL LENSES
-  - BROWN OR BROWNISH GRAY SILTY SAND
-  - GRAY SAND
-  - GRAY OR BROWNISH GRAY SILTY CLAY
-  - BROWN SILTY CLAY



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FIGURE 3-2 GEOLOGIC CROSS-SECTION A-A'



LEGEND









-  - GRAY SAND
-  - BROWN SAND WITH LOCALIZED SAND AND GRAVEL LENSES
-  - BROWN SILTY SAND
-  - BROWN SILT
-  - GRAY CLAYEY SILT
-  - BROWN SILTY CLAY
-  - DARK GRAY CLAY
-  - SHALE BEDROCK



FIGURE 2-3 GEOLOGIC CROSS-SECTION B-B'

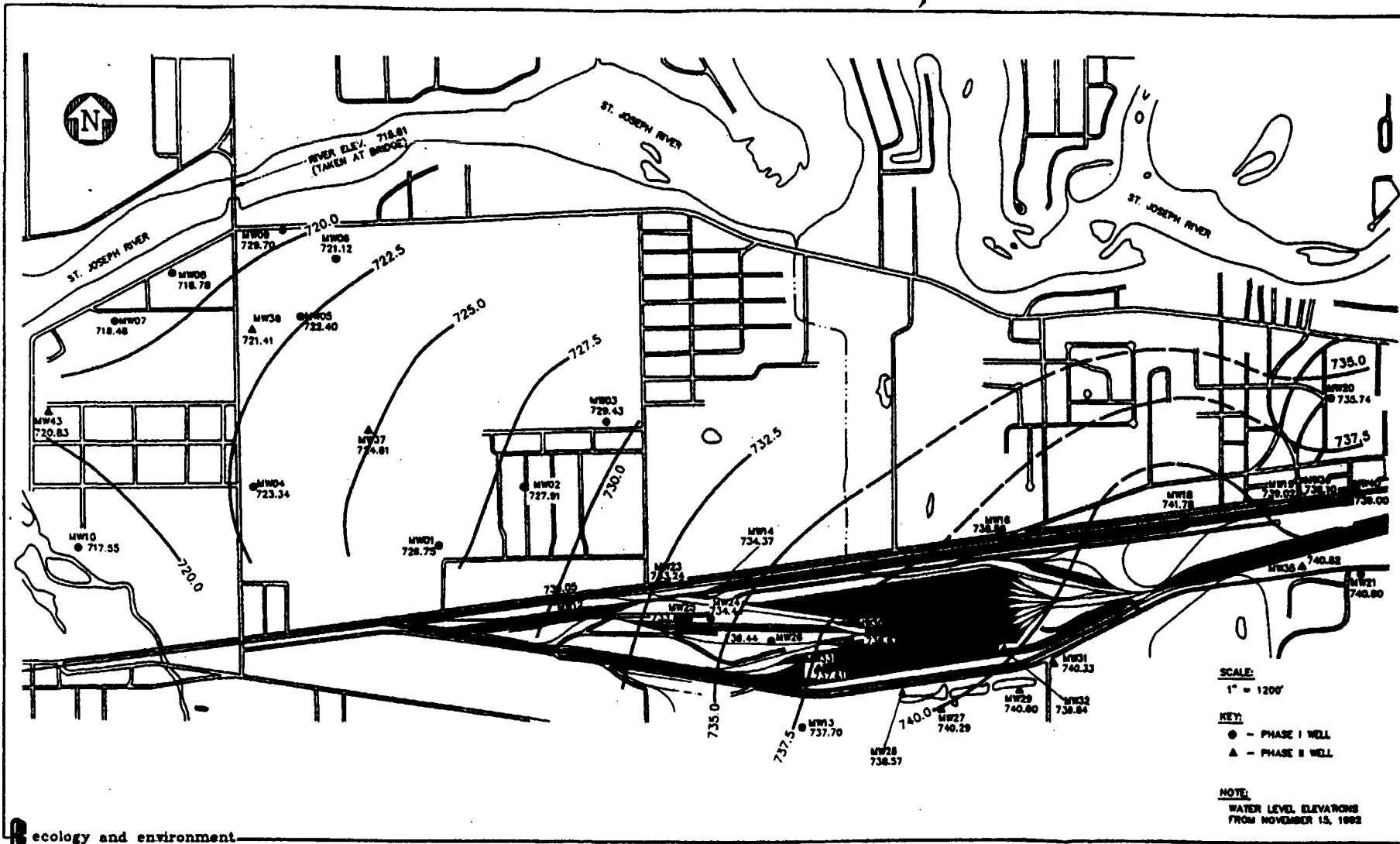


FIGURE 3-4
GROUNDWATER TABLE
SURFACE MAP FROM
NOVEMBER 1992 DATA

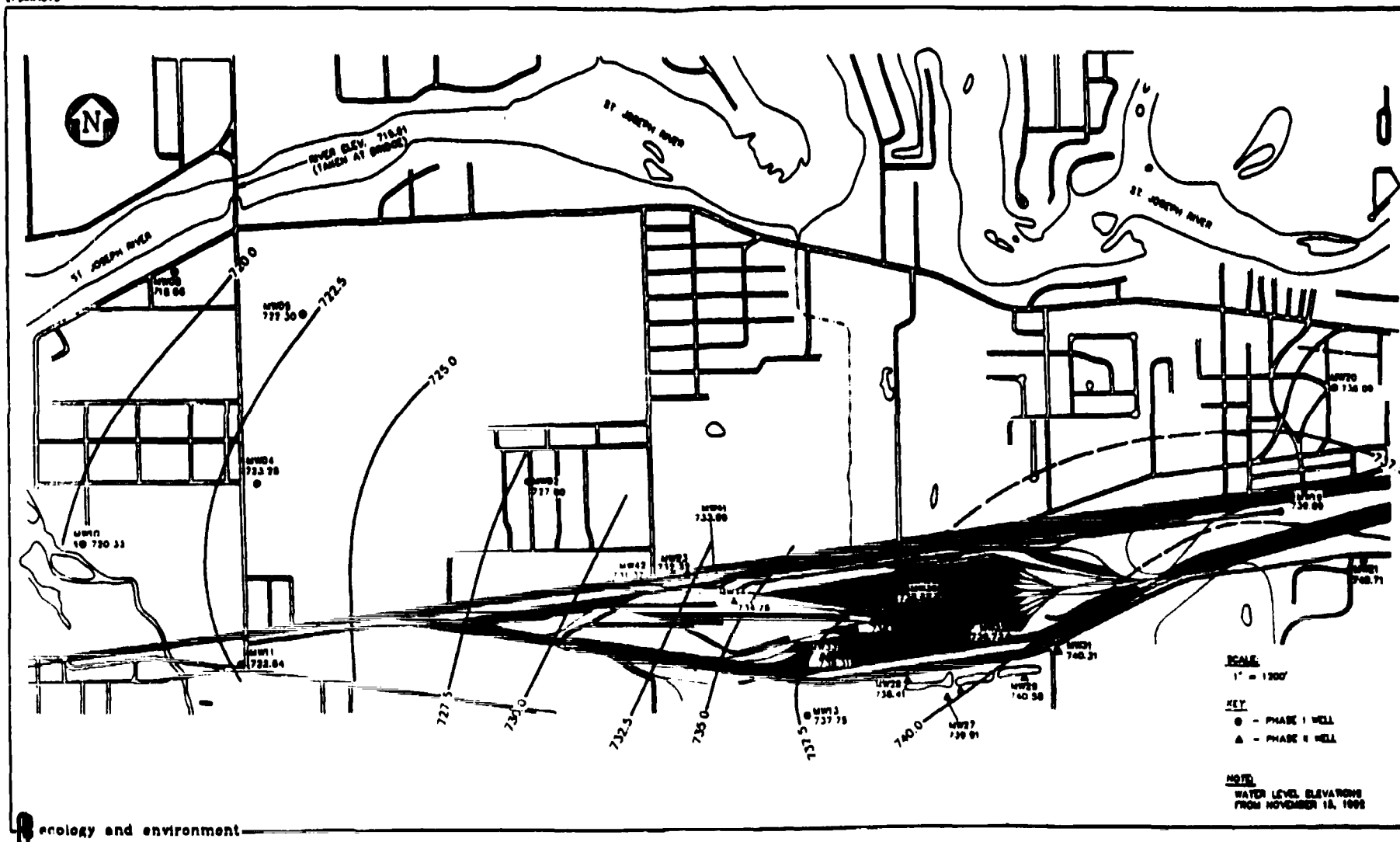


FIGURE 3-8 INTERMEDIATE GROUNDWATER POTENTIOMETRIC SURFACE MAP FROM NOVEMBER 1998 DATA

3-17

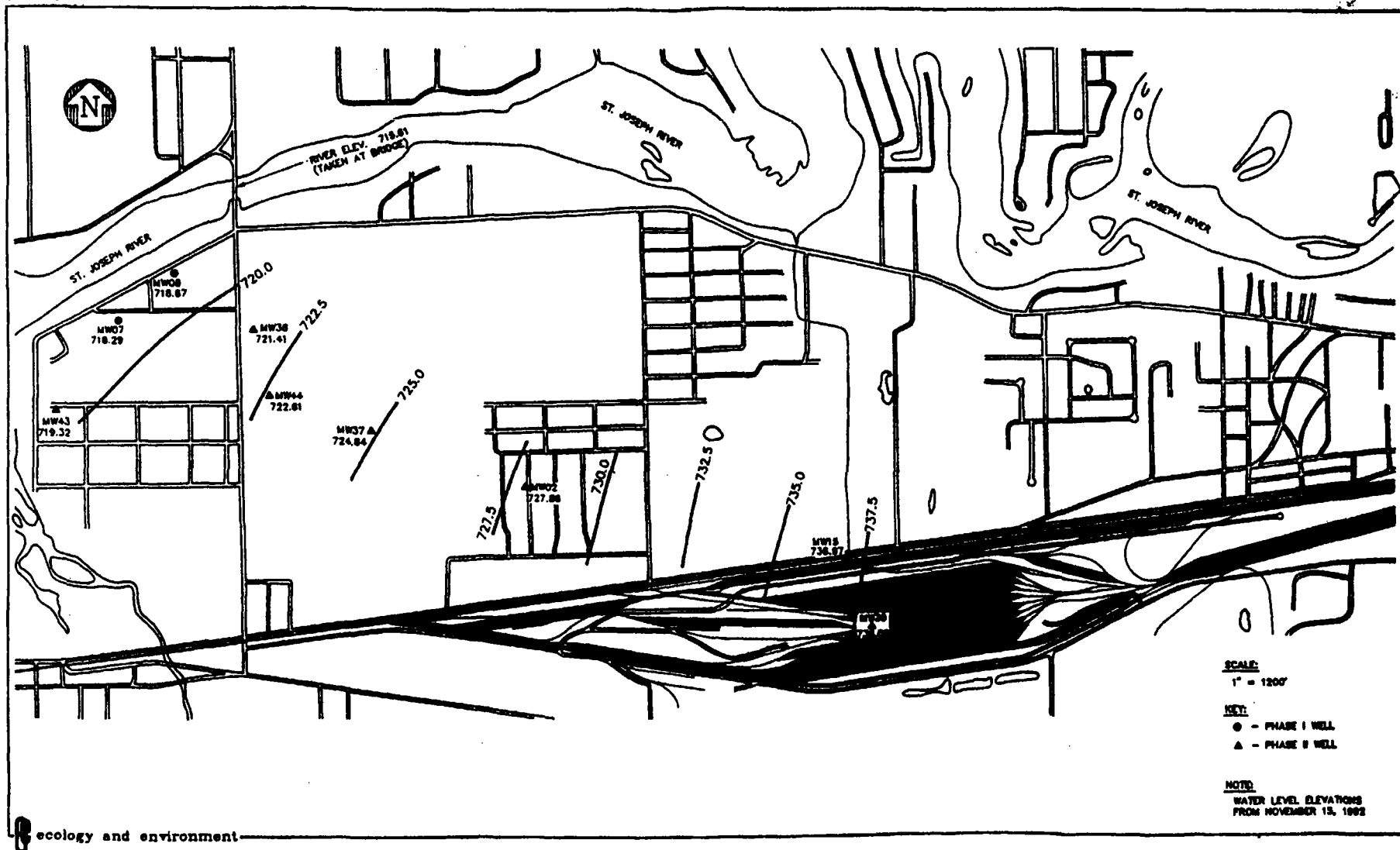


FIGURE 3-8

DEEP GROUNDWATER
POTENTIOMETRIC
SURFACE MAP FROM
NOVEMBER 1992 DATA

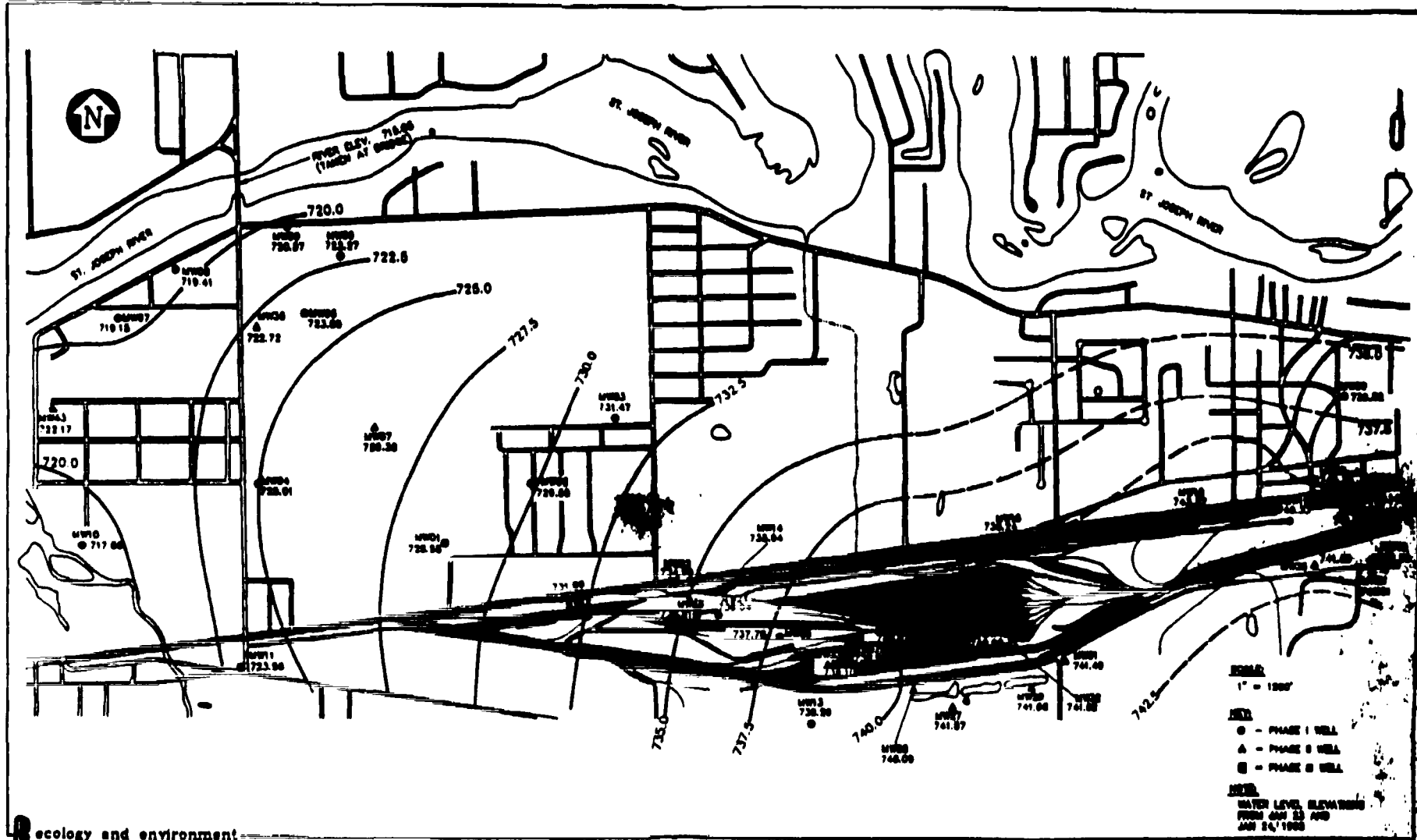


FIGURE 3-7

GROUNDWATER TABLE
SURFACE MAP FROM
JANUARY 1968 DATA

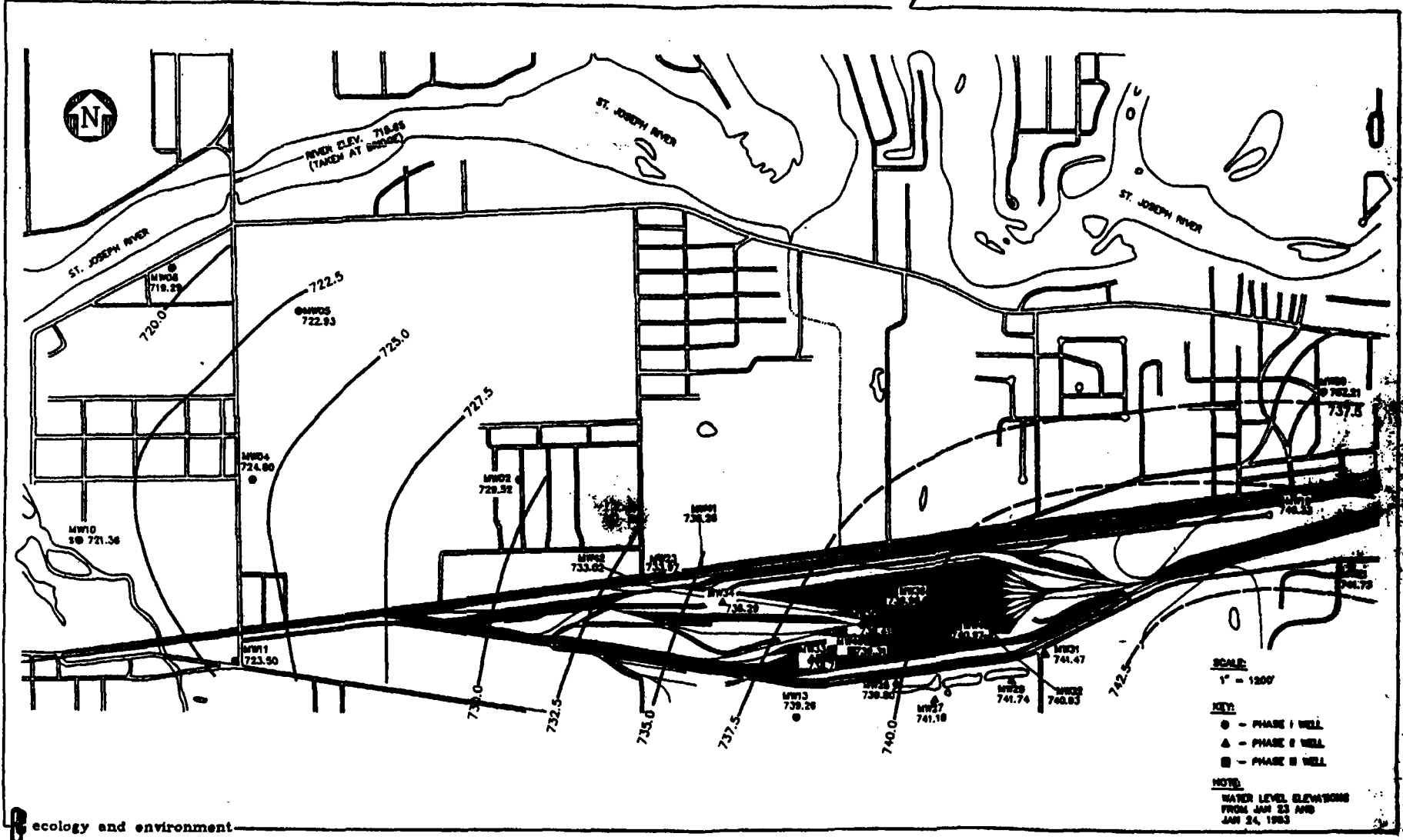
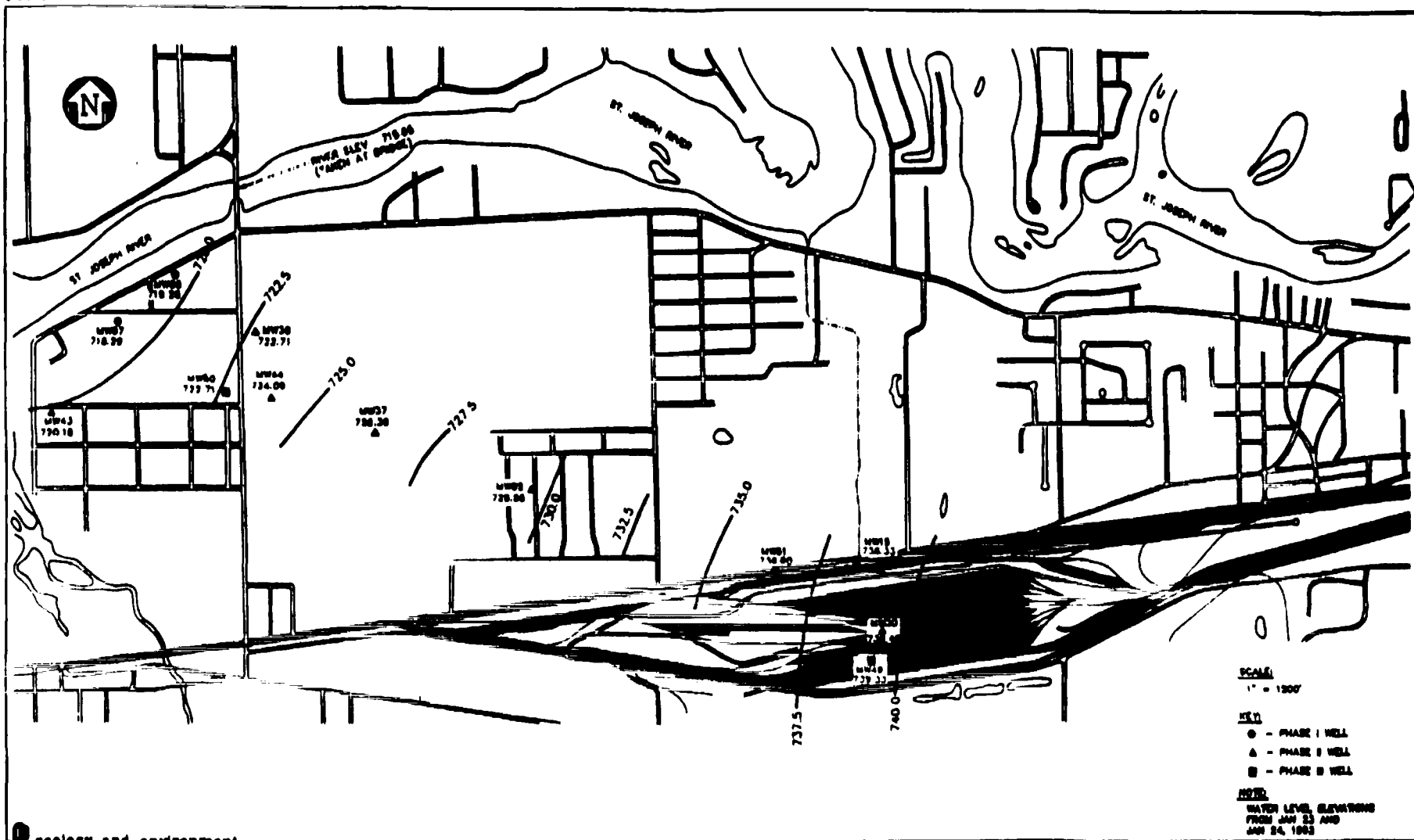


FIGURE 3-8

INTERMEDIATE GROUNDWATER
POTENTIOMETRIC SURFACE -
MAP FROM
JANUARY 1963 DATA



4. NATURE AND EXTENT OF CONTAMINATION

This section presents and discusses the nature and extent of contamination in soil, groundwater, and the drainage network. The discussion of the nature and extent of source areas is based on Phase I, II, and III soil analytical data.

4.1 LEAD-SCREEN AUGER SAMPLING RESULTS

This section summarizes and discusses the LSA sampling results from the Phase III field investigation. Eleven LSA borings were located in the County Road 1 plume and five LSA borings were located in the LaRue Street plume. All groundwater samples collected using the LSA technique were analyzed for CCl_4 , TCE, CHL, and TCA at the field laboratory. The method detection limits (MDLs) for these compounds were 8 $\mu\text{g/L}$, 9 $\mu\text{g/L}$, 7 $\mu\text{g/L}$, and 7 $\mu\text{g/L}$, respectively. Appendix C contains the LSA sampling results and Figure 2-1 shows the LSA boring locations. The vertical distribution of contamination is presented and discussed in a manner that is consistent with the interpretation of the groundwater flow patterns presented in Section 3.2 of this report. To facilitate the discussion of the groundwater monitoring approach and interpretation of analytical results, the aquifer has been divided into three depth intervals. These three zones are used as a means to evaluate site conditions and do not correspond to identifiable stratigraphic horizons. The shallow zone extends from the water table (approximately 15 feet BGS) to approximately 35 feet BGS. The intermediate zone spans the interval 35 feet BGS to 85 feet BGS. The deep zone extends from 85 feet BGS to the top of bedrock.

4.1.1 County Road 1 Plume

LSA boring LSA32 was placed in the track 69 CCl_4 source area, which was identified in the Phase II investigation, to define the vertical extent of groundwater contamination. This LSA boring was drilled and sampled to bedrock to determine the presence or absence of residual or pooled dense nonaqueous phase liquid (DNAPL) in localized horizons, including the top of bedrock. In the groundwater sample collected from the 18 to 23-foot BGS sample interval, CCl_4 was detected at 31,000 $\mu\text{g/L}$. This was the highest detected concentration of any contaminant in a groundwater sample collected using the LSA technique in the study area. CCl_4 concentrations in the two other samples collected between 23 feet and 35 feet BGS were 6,400 $\mu\text{g/L}$ and 1,800 $\mu\text{g/L}$. Within the intermediate monitoring zone, CCl_4 concentrations ranged from non-detect to 200 $\mu\text{g/L}$. CCl_4 was detected in groundwater samples collected from the deep zone at concentrations less than or equal to 84 $\mu\text{g/L}$. The groundwater sample collected from the interval on top of bedrock (143 to 148 feet BGS) shows detectable levels of CCl_4 ; however, this detection may be attributable to cross-contamination from the sampling pump.

LSA borings LSA34 and LSA36 were positioned at the approximate center of the main classification yard, which is hydraulically downgradient of LSA32, in order to locate the path of the CCl_4 groundwater plume originating from the track 69 source area. LSA34 and LSA36 were drilled to 93 feet and 63 feet, respectively. The highest concentration of CCl_4 detected in LSA34 was 2.2 $\mu\text{g/L}$ in the 23 to 28-foot sample interval, and the highest concentration of CCl_4 in LSA36 was 5.5 $\mu\text{g/L}$ in the 13 to 18-foot sample interval. This indicates that the CCl_4 contamination associated with the track 69 source area has limited impact on the groundwater quality at these locations.

TCE was detected in groundwater samples collected from LSA32, LSA34, and LSA36. TCE was detected only once in LSA32 above the MDL in the groundwater sample collected from the 28 to 33-foot sample interval at a concentration of 15 $\mu\text{g/L}$. TCE was detected above 1 $\mu\text{g/L}$ in LSA34 at concentrations of 5.7 $\mu\text{g/L}$ and 47 $\mu\text{g/L}$ in the groundwater samples collected from the 18 to 23-foot and 23 to 28-foot sample interval, respectively. At LSA36 located 250 feet east of LSA34, TCE was detected at a maximum concentration of 1,300 $\mu\text{g/L}$ in the groundwater sample collected from the 38 to 43-foot sample interval. TCE concentrations gradually decrease from this high concentration with depth in

the intermediate zone, such that TCE was detected at 25 $\mu\text{g/L}$ at the bottom of the boring at 63 feet BGS. Based on field analytical data from LSA34, which showed a maximum TCE concentration of 47 $\mu\text{g/L}$, and field analytical data from LSA36 and LSA25 (from Phase II, see Appendix B), located east and west of LSA34, respectively, each of which revealed TCE at concentrations greater than 1000 $\mu\text{g/L}$, it appears as if one TCE groundwater contamination plume has been bisected in the vicinity of LSA34, or multiple TCE plumes are commingled and are represented by analytical results from LSA36 and LSA25. The source(s) of the TCE detected in these borings is unknown.

TCA was detected in LSA32 at a concentration of 9.6 $\mu\text{g/L}$ in the groundwater sample collected from the 48 to 53-foot sample interval. TCA was either not detected or detected at concentrations $< 1 \mu\text{g/L}$ in all other samples from LSA32, LSA34, and LSA36. In these same borings, CHL is present in groundwater samples where CCl_4 is observed, at relatively lower concentrations. This is consistent with the degradation process of CCl_4 parent material undergoing reductive dechlorination to CHL daughter product (see Section 5.2.2).

LSA boring LSA38 was completed in the track 65 and 66 TCE source area, identified in the Phase II investigation, to define the vertical extent of the groundwater contamination in this area. The total depth of LSA38 is 148 feet BGS. LSA borings LSA42 and LSA44 are positioned downgradient of LSA38 in order to characterize the path of the plume originating from this source area. The total depths of LSA42 and LSA44 are 138 feet and 83 feet BGS, respectively. TCE was detected above the MDL concentration in groundwater samples collected from all three of these LSA borings, while CCl_4 , CHL, and TCA concentrations were detected below their respective MDLs in samples collected from the three locations. TCE concentrations from LSA38 ranged from 28 $\mu\text{g/L}$ in the 18 to 23-foot sample interval to 1,400 $\mu\text{g/L}$ in the 78 to 83-foot sample interval. TCE also was detected at a concentration of 720 $\mu\text{g/L}$ in the sample collected on top of bedrock (143 to 148 feet BGS) in LSA38. Concentrations of TCE greater than 500 $\mu\text{g/L}$ were present in the groundwater samples collected from the shallow monitoring zone of LSA42 with the highest concentration (2,800 $\mu\text{g/L}$) detected in the 33 to 38-foot sample interval, a sampling interval which overlaps the shallow and intermediate zones. TCE was not detected above the MDL in the deep monitoring zone in LSA42. In LSA44, TCE was detected at concentrations greater than or equal to

100 $\mu\text{g/L}$ between 18 feet BGS and 58 feet BGS, with a zone of TCE detected at concentrations greater than 500 $\mu\text{g/L}$ between 28 and 43 feet BGS. The results from LSA44 indicate that TCE contamination extends into the intermediate monitoring zone at this location. The shallow and intermediate contamination in this area can be attributed to the track 65 and 66 TCE source area. The presence of DNAPL or an additional source may be the cause of contamination in the deep zone, however the source of the deep zone contamination is unknown.

Three LSA borings, LSA43, LSA45, and LSA46, were positioned east to west, respectively, along U.S. 33 to identify the path of the plume(s) at the border of the railyard and to further define the groundwater contamination previously identified to the west of these locations in the intermediate monitoring zone. The total depths of these borings were 68 feet, 93 feet, and 108 feet, respectively. CCl_4 and TCA were not detected in samples collected from these three borings at concentrations above the respective MDL for each compound. CHL was detected only once above the MDL in groundwater samples collected from these three borings at a concentration of 38 $\mu\text{g/L}$ in the 103 to 108-foot sample interval in LSA46. TCE was detected above the MDL in groundwater samples collected from LSA43 at concentrations of 12 $\mu\text{g/L}$ and 13 $\mu\text{g/L}$ in the 23 to 28-foot and 63 to 68-foot sample intervals, respectively. In LSA45, TCE was detected above the MDL in the 58 to 63-foot and 78 to 83-foot sample intervals at respective concentrations of 350 $\mu\text{g/L}$ and 13 $\mu\text{g/L}$. In LSA46, TCE was not detected above the MDL in groundwater samples collected in the shallow and intermediate zones. In the groundwater samples collected within the deep zone, the TCE concentrations ranged from 1,400 $\mu\text{g/L}$ to 6,300 $\mu\text{g/L}$. Based on these data and Phase II LSA results, this TCE at the border of the Conrail facility represents a continuous plume that can be traced upgradient into the railyard and downgradient into the County Road 1 area.

LSA40 and LSA41 were located and sampled to investigate the relationship between the County Road 1 plume and the Charles Avenue contamination (see Figure 2-1). In LSA 40, CCl_4 and TCE were not detected at concentrations above the respective MDL for each compound in groundwater samples collected to a depth of 93 feet. CCl_4 , however, was detected in the deep zone at a concentration of 84 $\mu\text{g/L}$ in the groundwater sample collected from the 98 to 103-foot depth interval. CCl_4 concentrations increased to approximately 200

$\mu\text{g/L}$ at a depth of 108 feet and remained at this approximate concentration to a depth of 133 feet, before decreasing to concentrations below the MDL in the groundwater sample collected from the 143 to 148-foot sample interval at the top of bedrock. In addition, TCE was detected in groundwater samples collected from LSA40 in the five sample intervals between 108 and 133 feet BGS at concentrations ranging from 55 $\mu\text{g/L}$ to 98 $\mu\text{g/L}$. TCE was not detected above the MDL in groundwater samples collected between 133 feet and 148 feet BGS.

In LSA41, CCl_4 was not detected above the MDL in groundwater samples collected to a depth of 146 feet (top of bedrock). TCE, however, was initially detected above the MDL in LSA41 in the groundwater sample collected from 108 feet BGS, at a concentration of 17 $\mu\text{g/L}$. TCE was detected at a concentration of 530 $\mu\text{g/L}$ in the groundwater sample collected from the 128 to 133-foot sample interval. TCE concentrations remained above 100 $\mu\text{g/L}$ to the top of bedrock at a depth of 146 feet. These results indicate that the Charles Avenue contamination is part of the County Road 1 plume. Within the Charles Avenue area, the areal extent of TCE groundwater contamination appears to be greater than the CCl_4 groundwater contamination.

4.1.2 LaRue Street Plume

LSA borings LSA31, LSA33, LSA35, LSA37, and LSA39 were drilled in the east end of the Conrail Site study area, on the Conrail facility adjacent to the receiving yard, in order to delineate the source of the LaRue Street contamination plume within the railyard (see Figure 2-1). Based on data from previous investigations in the study area, the levels of contamination are lower in the LaRue Street area than those observed in the main classification yard. Levels of contamination in this area are near or less than the respective MDLs. Figure 4-1 shows the locations of cross sections C-C' and D-D' created from the LSA borings in the LaRue Street area. Figures 4-2 and 4-3 are cross sections C-C' and D-D', respectively, and show the concentration contours of CCl_4 and TCA within the aquifer.

Cross-section C-C' (see Figure 4-2) is oriented south to north and includes LSA borings LSA31, LSA35, and LSA33, respectively. The Conrail receiving yard is approximately midway between LSA31 and LSA35. The groundwater flow direction is approximately to the north. LSA31 is upgradient of the receiving yard based on groundwater flow

direction and has a total depth of 93 feet BGS. The highest concentration of any of the four volatile compounds in groundwater samples collected from this boring analyzed for in the field laboratory was 3.5 $\mu\text{g/L}$ TCA in sample interval 53 to 58 feet BGS. This indicates that, with the exception of TCA, there are no sources of these compounds immediately upgradient of the receiving yard contributing to the groundwater contamination identified downgradient of this location. The location of the source of this TCA is not known. TCA can be traced within the intermediate zone with increasing concentrations downgradient of the receiving yard from LSA35 to LSA33. TCA was not detected in groundwater samples collected from the deep zone. The source of this TCA also is not known.

Cross section C-C' (Figure 4-2) shows the absence of CCl_4 in LSA31 near the water table and the presence of CCl_4 at 13 $\mu\text{g/L}$ in the water table groundwater sample collected from the 18 to 23-foot interval of LSA35, which is located downgradient of the receiving yard on the Conrail facility. Further downgradient in the general direction of groundwater flow, CCl_4 was detected in LSA33 at a concentration of 34 $\mu\text{g/L}$ in the groundwater sample collected from the 18 to 23-foot sample interval and at a concentration of 24 $\mu\text{g/L}$ in the groundwater sample collected from the 23 to 28-foot sample interval. The results shown in the C-C' cross section are consistent with the Phase II LSA results. Specifically, LSA10, which was completed at the present location of MW40, was sampled during the Phase II RI and revealed CCl_4 concentrations similar to those at LSA33 at a similar depth. CCl_4 was not detected above the MDL in groundwater samples collected from the intermediate or deep groundwater zones.

The line of section for cross section D-D' (Figure 4-3) is oriented east to west and includes LSA borings LSA39, LSA35, and LSA37, respectively. The LSA borings used to create this cross section are all located downgradient of the receiving yard. LSA37 was located 30 feet west of LSA35, and LSA39 was located 50 feet east of LSA35. The shallowest sample interval, 18 to 23 feet BGS, from each of these three LSA borings exhibits the highest concentration of CCl_4 for the respective boring. For this sample interval, CCl_4 was detected at concentration of 5.6 $\mu\text{g/L}$ in LSA37, at a concentration of 13 $\mu\text{g/L}$ in LSA35, and at a concentration of 70 $\mu\text{g/L}$ in LSA39. Concentrations rapidly decrease with depth from these high concentrations. The western edge of this CCl_4 plume seems fairly well defined by the groundwater results from LSA35 and LSA37, yet the eastern edge of the

plume as identified through groundwater results from LSA39 indicates that CCl_4 groundwater contamination extends further east and remains uncharacterized.

TCA groundwater contamination is limited to the intermediate zone in LSA37 and LSA35. The shallow and intermediate zones, however, of LSA39 reveal TCA groundwater contamination. The deep zones for these borings were not investigated. The pattern of TCA contamination shown in Figure 4-3 is distinctly different from the pattern of CCl_4 contamination, suggesting separate contributing sources for these two compounds.

4.2 SOIL SAMPLING RESULTS

This section summarizes the subsurface soil sample analytical results from the Phase III soil borings. Appendix D contains the Phase III soil boring logs, and the analytical results for the Phase III subsurface soil samples are presented in Appendix E, Table E-1. The discussion portion of this section is an interpretation of the nature and extent of soil contamination based on subsurface soil data from all three phases of investigation.

Five distinct areas were targeted in the Phase III subsurface soil investigation. Figure 4-4 presents the locations of the Phase III soil borings along with selected volatile organic results for subsurface soil samples. Two of the areas were sampled to further define the extent of sources identified during the Phase II field investigation: the track 69 area in the eastern end of the classification yard and the track 65 and 66 area in the western end of the classification yard. All soil samples collected from these two areas were analyzed only for TCL VOCs at a CLP laboratory. Three additional areas were investigated to determine whether they were source areas potentially contributing to known groundwater contamination: an area in the receiving yard upgradient of the LaRue Street area groundwater contamination, an area of reported spilled drums located south of the repair yard and east of the car shop, and a location where a tank car was reportedly buried. Soil samples collected from these three areas were analyzed for TCL VOCs, semivolatiles, pesticides/PCBs, and TAL metals at a CLP laboratory. Semivolatiles, pesticides/PCBs, and metals sample analyses will be used during the FS, if necessary, and are also presented in Appendix E, Table E-1. Soil boring locations were selected based upon LSA results where applicable; Phase II RI results; and site background information. Soil sample intervals were selected for VOC analysis based on OVA readings in split-spoon samples (presented in Appendix D) and/or based on the depth and

aquifer material relative to the LSA groundwater results. Table 2-1 presents the sampling and analysis program for Phase III subsurface soils and indicates the sample interval sent for analysis.

4.2.1 Track 69 Source Area

Three soil borings, B-40, B-41, and B-42, were completed in the CCl_4 track 69 source area in order to define the north and south extent of the source area. B-40, located immediately north of track 68 in this source area, was advanced to bedrock in order to define the vertical extent of the CCl_4 source and to investigate the potential for residual and/or pooled dense non-aqueous phase liquid (DNAPL). Soil borings B-41 and B-42 were located one track north and two tracks south, respectively, of the B-40 location. In soil boring B-40, CCl_4 was detected in sample intervals of 18 to 20 feet BGS and 22 to 24 feet BGS, at concentrations of 1,100 $\mu\text{g/kg}$ and 64,000 $\mu\text{g/kg}$, respectively. CCl_4 was detected again in this soil boring in the 128 to 130 feet BGS sample interval at a concentration of 16 $\mu\text{g/kg}$. CCl_4 was not detected in the 148 to 150 foot BGS sample interval, which is located at the top of bedrock. CCl_4 was detected to a depth of 25 feet BGS in B-41 and B-42, at respective concentrations of 4,200 $\mu\text{g/kg}$ and 19 $\mu\text{g/kg}$. Soil sample intervals from soil borings in this area were selected for analysis based on LSA groundwater results.

4.2.2 Track 65 and 66 Source Area

Three soil borings, B-47, B-51, and B-52, were completed in the TCE source area between tracks 65 and 66 in the western portion of the classification yard, in order to define the north and south extent of this source area. In soil boring B-47, the northernmost boring, TCE was detected at 70 $\mu\text{g/kg}$ in the 2 to 4 foot BGS sample interval. In soil boring B-51, TCE was detected in the 2 to 4-foot BGS and 4 to 6-foot BGS sample intervals, at respective concentrations of 20 $\mu\text{g/kg}$ and 89 $\mu\text{g/kg}$. The compound 1,2-dichloroethene (total), a degradation product of TCE, was detected in sample interval 2 to 4 feet BGS at a concentration of 1,100 $\mu\text{g/kg}$. In the southernmost boring, B-52, TCE was not detected in the soil samples collected.

4.2.3 Other Potential Source Areas

Three soil borings, B-43, B-44, and B-45, were completed in the field just south of the repair yard where drum contents were reportedly spilled. TCE was not detected in soil boring B-43. TCE was detected in soil boring B-44 at a concentration of 8 $\mu\text{g/kg}$ from the 15 to 17-foot BGS sample interval, and in soil boring B-45 at a concentration of 17 $\mu\text{g/kg}$ from the 23 to 25-foot BGS sample interval. Soil samples collected from the 7 to 9-foot BGS intervals from soil borings B-44 and B-45 did not detect TCE. These detected concentrations of TCE were detected in soil sample intervals at or near the water table. CCl_4 was not detected in any of the soil samples collected from these borings. Acetone was also detected in soil borings B-43 and B-44 at a maximum concentration of 230 $\mu\text{g/kg}$. Although acetone is a common laboratory artifact, it was also detected at low, unqualified concentrations of 38 $\mu\text{g/L}$ and 19 $\mu\text{g/L}$ in groundwater samples collected from monitoring wells MW41 and MW51, respectively.

The reported buried tank car was located under a concrete slab adjacent to the northwest corner of the Bridges and Buildings (B and B) Shop. Because the tanker was empty and not filled with soil as anticipated, no samples from inside the tanker could be collected. A soil boring, B-46, was completed, however, north-northwest of the tanker at a hydraulically downgradient location, in order to determine whether the former contents of the tanker had leaked into the subsurface and were contributing to groundwater contamination. Acetone was detected in the 19 to 21-foot BGS sample interval at a concentration of 41 $\mu\text{g/kg}$. Three soil borings, B-48, B-49, and B-50, were completed in the receiving yard on the Conrail facility to identify the source(s) of the LaRue Street contamination plume. The soil boring locations were determined based on LSA groundwater results. In soil borings B-48 and B-49, CCl_4 was detected in soil samples collected just above the water table at concentrations of 31 $\mu\text{g/kg}$ and 22 $\mu\text{g/kg}$, respectively. In soil boring B-50, CCl_4 was detected at a concentration of 29 $\mu\text{g/kg}$ in the 0 to 2-foot BGS sample interval.

4.2.4 Discussion

Soil borings, along with subsurface soil sample collection, were completed during each of the three phases of field investigation in order to determine the nature and extent of identified and suspected source areas contributing to identified groundwater contamination.

Figure 4-5 presents the soil boring locations. Soil borings B-01 through B-19 were completed during the Phase I field investigation. Soil borings B-12, B-13, and B-14 are located east of the Conrail facility in the East Leaking Tank Car area, which is outside of the study area as it is presently defined; and therefore, they are not presented on Figure 4-5. Soil borings B-20 through B-39 were completed during the Phase II field investigation, and B-40 through B-52 were completed during the Phase III field investigation. Based on analytical results from subsurface soil samples, two source areas on the Conrail facility have been identified. A discussion of each of the source areas follows.

A CCl_4 source area was identified in the eastern section of the classification yard based on subsurface soil samples from soil borings B-24, B-25, B-26, B-40, B-41, and B-42. Soil borings B-24, B-25, and B-26 are positioned west to east, respectively, in a 115-foot line that is parallel to and just north of track 69. Seventy-five feet separate B-24 (western boring) and B-25, and 40 feet separate B-25 and B-26 (eastern boring). Soil borings B-40, B-41, and B-42 are positioned on a 30-foot north-south-trending line between tracks 67 and 70, roughly bisecting the B-25 location. Based on analytical data from soil samples collected from these borings, CCl_4 contamination was detected in an area bounded on the west and east by B-24 and B-25, respectively (75 feet), and on the north and south by B-41 and B-42, respectively, (30 feet). CCl_4 contamination was detected in soil samples collected from these borings between the depths of 18 feet BGS and 25.5 feet BGS (7.5 feet). The analytical data from these boundary locations are greater than or equal to 1 mg/kg, suggesting that this CCl_4 source area extends beyond the approximate boundaries established with the data to date. This source is located in the saturated zone, in a stratigraphic unit that is more silty than the stratigraphic units above and below it. B-40 was drilled to the top of bedrock (150 feet) and soil samples were collected throughout the length of borehole. CCl_4 was detected only once between 58 feet and 150 feet BGS at 16 $\mu\text{g/kg}$ in the 128 to 130-foot interval sample. Chloroform was also detected in this interval at a concentration of 9 $\mu\text{g/kg}$. Groundwater data (see Section 4.3) and site background information indicate the presence of a CCl_4 DNAPL source.

A TCE source area was identified in the western section of the classification yard, approximately 1,900 feet west of the eastern straight-a-way between tracks 65 and 66. Approximate dimensions of this source area are based on analytical data from subsurface soil

samples collected from soil borings B-27, B-28, B-29, B-32, B-47, B-51, and B-52. Soil borings B-27, B-28, B-29, and B-32 are positioned on a 440-foot east-west line just north of track 66. B-28 is located at the 1,900-foot mark west of the eastern straight-a-way; B-29 is 40 feet to the west of B-28; and B-32 and B-27 are 80 feet and 400 feet, respectively, to the east of B-28. Soil borings B-47, B-51, and B-52 are positioned north to south, respectively, in a 16-foot line between tracks 65 and 67, roughly bisecting the B-28 location. TCE contamination was detected in an area bounded on the west and east by B-29 and B-32, respectively (120 feet), and on the north and south by B-47 and B-28 (10 feet). TCE contamination was detected in soil samples collected from these soil borings at depths from 0 feet to 4 feet BGS. This TCE source area is located in the unsaturated zone. However, groundwater data from wells located directly downgradient from this source (MW49D and MW49BR) detect TCE contamination at depths much greater than 4 feet BGS, indicating unidentified TCE contamination deeper in the subsurface or an unidentified DNAPL TCE source.

CCl_4 contamination was detected in subsurface soil samples collected from soil borings B-48, B-49, and B-50 located on an east-west-trending line, just north of track 6 in the receiving yard at the eastern end of the site, on the Conrail facility in the LaRue Street area. Soil boring B-48 is located 970 feet west of the Conrail bridge over Nappanee Street, and B-49 and B-50 are 50 feet and 100 feet west, respectively, from B-48. The contamination in soil samples in this area is of low concentration ($\leq 31 \mu\text{g/kg CCl}_4$), yet the CCl_4 groundwater contamination in the LaRue Street plume is also of low concentration. CCl_4 was detected in the 0 to 2-foot BGS sample interval in soil boring B-50. Although the concentrations of CCl_4 detected in the soil samples do not definitely define a source area, they do indicate the presence of surface and subsurface CCl_4 contamination potentially contributing to groundwater contamination.

4.3 GROUNDWATER MONITORING WELL SAMPLING RESULTS

This section summarizes and discusses the results of groundwater samples collected from monitoring wells during the Phase III field investigation. The County Road 1 plume will be presented first, followed by the LaRue Street plume. Phase III groundwater analytical results are presented in Appendix E, Table E-2. All groundwater samples were analyzed for

TCL VOCs at a CLP laboratory. Phase III groundwater sample results are presented on Plate 1.

4.3.1 County Road 1 Plume

CCl_4 and TCE contamination were detected in groundwater samples collected from monitoring wells screened in the shallow, intermediate, and deep zones on the Conrail facility, at the St. Joseph River, and in the area between these points (see Plate 1). The contamination in Charles Avenue and Vistula Avenue are part of the single, indivisible County Road 1 plume. Groundwater flow direction in all three zones is west-northwest, and the groundwater contamination plume as defined by groundwater samples follows this path from the Conrail facility to the St. Joseph River. Based on groundwater flow direction, groundwater samples collected from monitoring wells located hydraulically upgradient of the plume and on the Conrail facility, MW27S and I, MW28S and I, MW29S and I, and MW31S and I, and groundwater samples from monitoring wells MW13S and D, located upgradient of the plume and the site, did not detect any VOC contamination. The identified CCl_4 and TCE groundwater contamination will be presented, respectively, for the shallow, intermediate, and deep monitoring zones of the aquifer.

The maximum concentration of CCl_4 groundwater contamination ($110,000 \mu\text{g/L}$) was detected in the shallow zone in a groundwater sample collected from monitoring well MW46S located in the track 69 area on the Conrail facility. Groundwater samples collected from shallow monitoring wells MW47, MW26, and MW24, all on the Conrail facility, also detected CCl_4 at concentrations ranging from $24 \mu\text{g/L}$ (MW26) to $150 \mu\text{g/L}$ (MW24). Downgradient of the Conrail facility, CCl_4 contamination was detected in groundwater samples collected from shallow monitoring wells MW14, MW02S, MW37S, MW38S, and MW08S, at concentrations ranging from $2 \mu\text{g/L}$ (MW14) to $820 \mu\text{g/L}$ (MW08S).

CCl_4 contamination was detected in the intermediate zone on the Conrail facility in groundwater samples collected from MW32I, MW46I, and MW49D at concentrations ranging from $1 \mu\text{g/L}$ (MW32I and MW49D) to $3 \mu\text{g/L}$ (MW46I). Groundwater samples collected from intermediate monitoring wells located directly downgradient of the Conrail facility, MW42, MW23D, and MW41, detected CCl_4 at respective concentrations of $150 \mu\text{g/L}$, $12 \mu\text{g/L}$, and $8 \mu\text{g/L}$. CCl_4 was also detected in groundwater samples collected from

intermediate monitoring wells located within the plume; MW02D, MW05D, and MW08D; at concentrations ranging from 17 $\mu\text{g/L}$ (MW05D) to 360 $\mu\text{g/L}$ (MW08D).

On the Conrail facility, CCl_4 was detected in the deep zone in a groundwater sample collected from MW49BR, screened on top of bedrock, at a concentration of 47 $\mu\text{g/L}$. CCl_4 was detected directly downgradient of the Conrail facility in a groundwater sample collected from a deep monitoring well, MW51, at a concentration of 42 $\mu\text{g/L}$. Within the plume downgradient of the Conrail facility, CCl_4 was detected in groundwater samples collected from the following deep monitoring wells: MW37D, MW38D, MW44, MW50, MW07D, and MW08BR. CCl_4 concentrations in these samples ranged from 42 $\mu\text{g/L}$ (MW08BR) to 3,200 $\mu\text{g/L}$ (MW38D).

TCE contamination was detected in the shallow zone on the Conrail facility in groundwater samples collected from monitoring wells MW46S, MW32S, MW47, MW30S, and MW24 at concentrations ranging from 1 $\mu\text{g/L}$ (MW46S) to 1,100 $\mu\text{g/L}$ (MW47). Shallow monitoring wells located directly downgradient of the Conrail facility detected TCE in the groundwater samples at concentrations of 13 $\mu\text{g/L}$ (MW14) and 75 $\mu\text{g/L}$ (MW23S). Groundwater samples collected from shallow monitoring wells MW02S, MW37S, MW38S, and MW08S, located within the plume, detected TCE at concentrations ranging from 16 $\mu\text{g/L}$ (MW37S) to 110 $\mu\text{g/L}$ (MW08S).

On the Conrail facility, TCE was detected in groundwater samples collected from the following intermediate monitoring wells: MW46I, MW32I, MW36, MW49D, MW30I, and MW34. Concentrations in these samples ranged from 8 $\mu\text{g/L}$ (MW49D) to 7,900 $\mu\text{g/L}$ (MW30I). Groundwater samples collected from intermediate monitoring wells located directly downgradient of the Conrail facility, MW42, MW23D, and MW41, had respective TCE concentrations of 150 $\mu\text{g/L}$, 64 $\mu\text{g/L}$, and 15,000 $\mu\text{g/L}$. The TCE concentration of 15,000 $\mu\text{g/L}$ detected in the groundwater sample collected from MW41 is the maximum TCE concentration detected in groundwater on the site. TCE concentrations detected in intermediate monitoring wells located within the plume, MW02D and MW08D, had respective concentrations of 500 $\mu\text{g/L}$ and 180 $\mu\text{g/L}$.

TCE was detected in a groundwater sample collected from a deep monitoring well, MW49BR, located on the Conrail facility, at a concentration of 20 $\mu\text{g/L}$. In the groundwater sample collected from deep monitoring well MW51, located directly downgradient of the

Conrail facility, TCE was detected at a concentration of 2,400 $\mu\text{g/L}$. TCE was detected in groundwater samples collected from the following deep monitoring wells located within the plume: MW02BR, MW37D, MW44, MW50, MW08BR, MW07D, and MW43BR. The TCE concentrations of the samples ranged from 5 $\mu\text{g/L}$ (MW08BR) to 1,100 $\mu\text{g/L}$ (MW07D).

4.3.2 Discussion of the County Road 1 Plume

Monitoring wells MW46 and MW47 are located directly downgradient of the CCl_4 source located in the track 69 area and the TCE source located between tracks 65 and 66 area, respectively, in order to determine whether these sources were contributing to identified groundwater contamination. Groundwater samples collected from MW46S and MW47 detected CCl_4 at a concentration of 110,000 $\mu\text{g/L}$ and TCE at a concentration of 1,100 $\mu\text{g/L}$, respectively. Both CCl_4 and TCE, however, were detected in monitoring wells not located directly downgradient of the identified sources. For instance, CCl_4 was detected in groundwater samples collected from MW49, MW47, MW26, and MW24, all located on the Conrail facility downgradient of the TCE source. Also, TCE was detected in the groundwater sample collected from MW36, which is located on the Conrail facility and downgradient of the CCl_4 source. A high level of TCE contamination was also detected in MW30I, located on the Conrail facility, side-gradient of the identified TCE source. These groundwater data strongly imply that additional sources of CCl_4 and TCE exist on the Conrail facility that are as yet unidentified. The spatial distribution of the maximum and high concentrations of CCl_4 and TCE throughout the plume (laterally and vertically) also suggests multiple, non-simultaneous spill episodes.

Chloroform (CHL), a degradation (daughter) product of CCl_4 , was detected in groundwater samples collected from shallow, intermediate, and deep monitoring wells located throughout the plume. It was only detected in groundwater samples from monitoring wells that also detected CCl_4 . The ratio of CCl_4 to CHL, calculated from detected concentrations in groundwater samples, does not form a consistent pattern that correlates with CCl_4 concentration, distance from the Conrail facility, or depth zone of the aquifer.

Methylene chloride (MCI) is also a daughter product of CCl_4 and of CHL. MCI was detected only once in a groundwater sample (MW46S) at a concentration of 30 $\mu\text{g/L}$. This

detection of MCI was not qualified as a laboratory artifact based on CLP quality-control protocols.

CCl_4 and TCE are both DNAPL compounds. Concentrations of CCl_4 and TCE in groundwater samples are equal to or greater than 1% of their respective solubilities ($8.00 \text{ E}+02$ and $1.10 \text{ E}+03$), suggesting the presence of a DNAPL source. The groundwater sample collected from MW46S detected 13.8% of the solubility of CCl_4 , and the groundwater sample collected from MW41 detected 1.4% of the solubility of TCE. Therefore, the sources of the identified groundwater contamination may be DNAPL sources (see Section 5 for further discussion of DNAPL sources). The distribution of CCl_4 and TCE in the shallow, intermediate, and deep zones, close to sources on the Conrail facility and in downgradient areas toward the river also suggests the presence of a DNAPL source. Contamination in the deep-zone adjacent to the river may also have migrated to that depth under the influence of the downward vertical gradients over the distance from the source to the river, but deep-zone contamination on the Conrail facility due solely to the downward vertical gradient is unlikely and may indicate DNAPL.

The groundwater sample collected from deep monitoring well MW43BR, located next to Baugo Bay, detected TCE at a concentration of $450 \mu\text{g/L}$ but did not detect CCl_4 at all, although similar concentrations of CCl_4 are detected in deep monitoring wells MW50 and MW44 ($250 \mu\text{g/L}$ and $650 \mu\text{g/L}$, respectively). This contamination is indivisible from the County Road 1 plume. Deep groundwater flow may be influenced by the direction of flow of the river, and may flow laterally (or southwesterly) prior to discharging to the river. If the TCE detected in MW43BR originated from a spill that occurred earlier than the CCl_4 spill, the deep flow, influenced by the river's flow, may have had time to carry the TCE contamination to this location but may not have had the time to transport the CCl_4 contamination to this location yet.

4.3.3 LaRue Street Plume

CCl_4 contamination was detected only in groundwater samples collected from monitoring wells screened in the shallow zone in the LaRue Street plume. Groundwater samples from monitoring wells MW45 and MW48, both located on the Conrail facility, and

MW40, located directly downgradient of the facility, detected CCl_4 at the respective concentrations of 76 $\mu\text{g/L}$, 13 $\mu\text{g/L}$, and 22 $\mu\text{g/L}$.

TCE contamination was detected only once at concentrations above the Maximum Contaminant Level (MCL) for TCE (5 $\mu\text{g/L}$) in the LaRue Street plume. The groundwater sample collected from the shallow monitoring well MW20S detected TCE at 15 $\mu\text{g/L}$. TCE concentrations detected in groundwater samples from shallow monitoring wells MW35, MW45, and MW48 were all below the MCL. TCE was detected once in the intermediate zone in the groundwater sample collected from MW20D located in the LaRue Street residential area at a concentration of 4 $\mu\text{g/L}$. TCE was not detected in groundwater samples collected from the deep zone in the LaRue Street plume.

4.3.4 Discussion of the LaRue Street Plume

Groundwater flow direction in the LaRue Street plume area is generally to the north. A very slight groundwater mound exists in the shallow zone, as evidenced by January 1993 groundwater elevation data for MW45, MW35, MW48, and MW21S (see Table 2-3 and Figure 3-7).

Monitoring well MW45 is located directly downgradient of the CCl_4 subsurface and near-surface soil contamination detected in the receiving yard (see Section 4.2). MW48 was located presumably upgradient of the receiving yard based on hydrologic data prior to the Phase III field investigation. The groundwater elevation data from MW21S indicates that the MW21S location is unaffected by the mounding, and can be considered directly upgradient or background to the LaRue Street plume and the Conrail facility. CCl_4 was not detected in the groundwater sample collected from MW21S, while groundwater and subsurface soil CCl_4 contamination was detected on the Conrail facility. CCl_4 contamination detected in the groundwater sample collected from MW48 is probably the result of mounding of groundwater, in the immediate area, transporting contamination in the direction of groundwater flow away from the mound.

TCE contamination was detected in groundwater samples collected from shallow monitoring wells on the Conrail facility at concentrations below the MCL for TCE. Concentrations of 15 $\mu\text{g/L}$ and 4 $\mu\text{g/L}$ were detected in the shallow and intermediate zones, respectively, in monitoring wells MW20S and MW20D in the La Rue Street residential area,

downgradient of the Conrail facility. TCE was not detected directly upgradient of the Conrail facility in the shallow and intermediate zones, as shown by analytical results of groundwater samples collected from MW21S or MW21D. Based on groundwater analytical data collected upgradient and downgradient of the Conrail facility, the source of the TCE contamination is on the Conrail property, but is, as yet, unidentified.

4.4 DRAINAGE NETWORK SAMPLING RESULTS

The purpose of the drainage network sampling was to determine whether the drainage network is currently acting as or was historically a source of the widespread groundwater contamination identified across the site. The drainage network schematic layout was supplied to E & E by Conrail during the Phase III field investigation. Figure 2-4 presents the drainage lines, flow directions, and sample locations. Network flow directions and access points (manholes) were determined from the schematic layout and from discussions with facility employees. The flow path of the network essentially is toward the main north-south line, which drains directly into the secondary oil and water separator south of Crawford Ditch. This oil and water separator eventually discharges into Crawford Ditch. Sample locations were determined based on locations of known and potentially contributing sources, the flow path of the network from those sources, and the ability to locate and access the network via manholes.

All drainage network samples were analyzed for TCL VOCs at a CLP laboratory. The analytical results are presented in Appendix E, Tables E-3 and E-4. Analysis of the sediment and water samples from the drainage network revealed no detectable levels of TCE or CCl_4 . Six volatile organic compounds were detected: acetone, methylene chloride, benzene, toluene, ethylbenzene, and xylenes (total). The acetone and methylene chloride can be considered common laboratory artifacts given the concentrations that were present. The remaining four compounds were detected in sediment sample DS02 and water sample DW02, both collected from a location in the drainage network immediately upstream of the discharge into the secondary oil and water separator. These four compounds are commonly associated with gasoline or diesel fuel. These results show that the drainage network is not currently a source of the identified VOC groundwater contamination, which resulted from surface spills on the facility. The possibility that the drainage network historically acted as a conduit for

VOC contamination cannot be assessed due to the lack of previous sampling data from the network.

4.5 SURFACE WATER AND SEDIMENT SAMPLING RESULTS

This section summarizes the analytical results of the sediment and surface water samples collected during the Phase III investigation in support of the risk assessment. An analysis of the results is presented in Section 6. The analytical results are presented in Appendix E, Tables E-5 and E-6. Tables 4-1 and 4-2 summarize the organic and inorganic analytical results, respectively, for each area in which a group of sediment samples was collected. Using the same format as the sediment data, the surface water sample organic and inorganic analytical results, respectively, are summarized in Tables 4-3 and 4-4. All surface water and sediment samples were analyzed for VOCs, semivolatile compounds, pesticides/PCBs, and inorganic analytes, except for SW16, which was not analyzed for pesticides/PCBs.

Three background sediment samples and three background surface water samples, SD/SW01 through SD/SW03, were collected from Bango Creek, upstream of its discharge to Bango Bay. As shown in Table 4-1, the VOC and semivolatile compounds detected in these background sediment samples were toluene and two phthalates. All three compounds are common laboratory artifacts. No pesticides or PCBs were detected. As shown in Table 4-2, sixteen inorganic analytes were detected in these sediment samples; the maximum concentration of each analyte detected in this group of samples is also listed in the table. No VOCs were detected in this group of surface water samples (see Table 4-3). Two phthalates and one pesticide were the compounds detected in these surface water samples. The phthalate compounds are common laboratory artifacts. The twelve inorganic analytes that were detected in the surface water samples, along with the maximum concentration of each analyte detected within this group, are shown in Table 4-4.

Five sample pairs were collected from Bango Bay (SD/SW04 through SD/SW08), in the approximate area where the County Road 1 plume discharges into the bay. The VOCs detected in the sediment samples were toluene, acetone, and methylene chloride. One phthalate and seven polycyclic aromatic hydrocarbon (PAH) compounds were detected in this group of sediment samples. Three pesticide compounds and Aroclor-1254 were also detected in sediment samples from this group. Eighteen inorganic analytes were detected. The highest

concentrations of semivolatile compounds detected in the sediment samples collected from Baugo Bay were found in SD08. Most of these compounds were not detected in the background samples. Sediment sample SD08 also exhibited higher levels of most inorganic analytes than were present in the background samples. Although only the VOCs and the phthalate are common laboratory artifacts, it is not clear whether any of these compounds or analytes are site-related. No VOCs were detected in this group of surface water samples. One semivolatile compound, a phthalate, was detected and it is classified as a common laboratory artifact. No pesticide or PCB compounds were detected in this group of surface water samples. Twelve inorganic analytes were detected. Surface water sample SW05 exhibited higher concentrations of most inorganic analytes than were observed in the background samples. Given the nature of the compounds and analytes detected in the surface water and sediment samples from Baugo Creek and Baugo Bay, their relationship to the site is uncertain.

Six sediment/surface water sample pairs were collected as background samples from the St. Joseph River. These samples (SD/SW17 through SD/SW22) were collected upstream of the location where Crawford Ditch discharges to the St. Joseph River. The VOCs detected in this group of sediment samples were acetone, 2-butanone, and methylene chloride, which are classified as common laboratory artifacts. Eighteen semivolatile compounds (consisting of phenols, aromatics, phthalates, and PAHs), two pesticides, and Aroclor-1254 were detected. Seventeen inorganic analytes were detected. No VOCs were detected in the surface water samples. Four phthalate compounds, each classified as a common laboratory artifact, were detected. One pesticide was detected. Eleven inorganic analytes were detected.

Eight sediment samples and eight surface water samples (SD/SW09 through SD/SW16) were collected from the St. Joseph River in an area that is part of the discharge zone of the County Road 1 plume. Acetone, methylene chloride, toluene, 2-butanone, 1,2-dichloroethene(total), TCE, and CCl_4 were the VOCs detected in this group of sediment samples. The latter three compounds are attributable to the site and are not considered to be common laboratory artifacts. Some of these site-related compounds were detected in two of the eight sediment samples in this group. TCE and CCl_4 were detected in sediment sample SD10 at concentrations of 100 $\mu\text{g/kg}$ and 67 $\mu\text{g/kg}$, respectively. TCE and 1,2-dichloroethene(total) were detected in sediment sample SD15 at concentrations of 12 $\mu\text{g/kg}$

and 32 $\mu\text{g}/\text{kg}$, respectively. One phthalate, nine PAHs, three pesticides, and Aroclor-1254 were the other compounds detected in this group of sediment samples. Eighteen inorganic analytes were also detected in this group of sediment samples. The types of compounds and analytes as well as their concentrations were similar to the background sample results from the St. Joseph River. TCE was the only VOC detected in this group of surface water samples. TCE was detected at a concentration of 35 $\mu\text{g}/\text{L}$ in surface water sample SW15. The TCE in this sample may be attributable to the site. No VOCs were detected in SW10, the sample corresponding to SD10. Four phthalate compounds were the only semivolatile compounds detected in this group of surface water samples and each is regarded as a common laboratory artifact. No pesticides or PCBs were detected. Twelve inorganic analytes were detected. The sample locations for this group of samples correspond to the discharge zone of the plume. The relationship between the compounds and analytes detected in this group of samples and the site is uncertain because similar results were obtained for the background samples. Given the nature of these compounds and analytes, only TCE, CCl_4 , and 1,2-dichloroethene(total) can be attributed to the site through the findings of this investigation.

Three sediment samples and three surface water samples (SD/SW23 through SD/SW25) were collected from the three ponds located south of the main classification yard on the Conrail facility. Acetone and toluene, each categorized as a common laboratory artifact, were the only VOCs detected in the sediment samples. Eighteen semivolatile compounds were detected in this group of sediment samples, with most of the detections present in SD25, the sample collected from the westernmost pond where an outfall pipe is located. Eleven pesticides were detected in SD25 and none were detected in the other two sediment samples. Aroclor-1254 was detected in SD25 at a concentration of 2,000 $\mu\text{g}/\text{kg}$. Twenty inorganic analytes were detected. VOCs, semivolatile compounds, pesticides, and PCBs were not detected in these three surface water samples. Eleven inorganic analytes were detected. Many of the compounds and analytes detected in sediment sample SD25 appear to be related to the sample location, which is near the outfall pipe. The apparent impact of the outfall is uncertain because the way in which the outfall pipe is connected to the Conrail facility is unknown.

Table 4-1

SUMMARY OF ORGANIC DATA FOR SEDIMENT SAMPLES

Organic Parameter	Bongo Bay Background Sample Numbers SD01-SD03				Bongo Bay Sample Numbers SD04-SD06				Ponds Sample Numbers SD23-SD25			
	Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration	
Volatile Organic Compounds												
methylene chloride	ND	ND	0/3	N/A	ND	46	3/5	SD06	ND	ND	0/3	N/A
acetone	ND	ND	0/3	N/A	ND	100	3/5	SD06	ND	840	2/3	SD25
1,2-dichloroethane (total)	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
2-butanone (MEK)	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
carbon tetrachloride	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
trichloroethene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
toluene	ND	4	1/3	SD02	ND	15	1/5	SD05	ND	12	1/3	SD25
Semivolatile Organic Compounds												
4-methylphenol	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	310	1/3	SD25
naphthalene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	110	1/3	SD25
2-methylnaphthalene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	120	1/3	SD25
acenaphthene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	110	1/3	SD25
dibenzofuran	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	52	1/3	SD25
fluorene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	80	1/3	SD25
phenanthrene	ND	ND	0/3	N/A	ND	350	1/5	SD06	ND	390	1/3	SD25
anthracene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	82	1/3	SD25

Table 4-1

FOR SEDIMENT SAMPLES

Organic Parameter	Range Bay Background Sample Numbers SD01-SD03				Range Bay Sample Numbers SD04-SD06				Ponds Sample Numbers SD23-SD25			
	Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration	
Substituted Organic Compounds (Cont.)												
carbazole	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	60	1/3	SD25
fluoranthene	ND	ND	0/3	N/A	ND	630	3/3	SD08	ND	430	1/3	SD25
pyrene	ND	ND	0/3	N/A	ND	910	3/3	SD08	ND	2,300	1/3	SD25
butylbenzylphthalate	ND	89	1/3	SD01	ND	ND	0/3	N/A	ND	ND	0/3	N/A
benzo(a)anthracene	ND	ND	0/3	N/A	ND	430	2/3	SD08	ND	770	2/3	SD25
chrysene	ND	ND	0/3	N/A	ND	330	2/3	SD08	ND	960	2/3	SD25
bis(2-ethylhexyl)phthalate	ND	73	2/3	SD01	ND	1,200	3/3	SD08	ND	710	2/3	SD24
di-n-octylphthalate	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	ND	0/3	N/A
benzo(b)fluoranthene	ND	ND	0/3	N/A	ND	770	1/3	SD08	ND	1,100	1/3	SD25
benzo(k)fluoranthene	ND	ND	0/3	N/A	ND	390	1/3	SD08	ND	540	1/3	SD25
benzo(a)pyrene	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	810	1/3	SD25
indeno(1,2,3-cd)pyrene	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	750	1/3	SD25
benzo(g,h,i)perylene	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	ND	0/3	N/A
Furans/PCBs												
Aldrin	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	8.4	1/3	SD25
Endosulfan I	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	6.2	1/3	SD25

0108

4-22

0108

Table 4-1

SUMMARY OF ORGANIC DATA FOR SEDIMENT SAMPLES

Organic Parameter	Bouge Bay Background Sample Numbers SD01-SD03				Bouge Bay Sample Numbers SD04-SD08				Ponds Sample Numbers SD23-SD25			
	Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections	Location of Maximum Concentration	
Pesticides/PCBs (Cont.)												
Dieldrin	ND	ND	0/3	N/A	ND	4.1	2/5	SD05	ND	ND	0/3	N/A
4,4'-DDE	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	17.0	1/3	SD25
Endrin	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	17.0	1/3	SD25
Endosulfan II	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	1.2	1/3	SD25
4,4'-DDD	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	1.2	1/3	SD25
4,4'-DDT	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	220.0	1/3	SD25
Methoxychlor	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	11.0	1/3	SD25
Endrin Aldehyde	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	5.6	1/3	SD25
alpha chlordane	ND	ND	0/3	N/A	ND	1.6	1/5	SD04	ND	9.2	1/3	SD25
gamma chlordane	ND	ND	0/3	N/A	ND	3.0	1/5	SD08	ND	15.0	1/3	SD25
Aroclor-1254	ND	ND	0/3	N/A	ND	130.0	1/5	SD08	ND	2,000.0	1/3	SD25

Table 4-1
SUMMARY OF ORGANIC DATA FOR SEDIMENT SAMPLES

Organic Parameter	St. Joseph River Background Sample Numbers SD17-SD22				St. Joseph River Sample Numbers SD09-SD16			
	Results Range (µg/kg)	Number of Detections*	Location of Maximum Concentration		Results Range (µg/kg)	Number of Detections*	Location of Maximum Concentration	
Volatile Organic Compounds								
methylene chloride	ND	10	2/6	SD17/21	ND	39	5/8	SD14
acetone	ND	230	2/6	SD17	ND	33	2/8	SD09/11
1,2-dichloroethane (total)	ND	ND	0/6	N/A	ND	32	1/8	SD15
2-butanone (MEK)	ND	15	2/6	SD19/22	ND	44	1/8	SD14
carbon tetrachloride	ND	ND	0/6	N/A	ND	67	1/8	SD10
trichloroethene	ND	ND	0/6	N/A	ND	100	2/8	SD10
toluene	ND	ND	0/6	N/A	ND	15	2/8	SD12
Semivolatile Organic Compounds								
4-methylphenol	120	120	1/1	SD19	ND	ND	0/7	N/A
naphthalene	N/A	N/A	0/0	N/A	ND	ND	0/7	N/A
2-methylnaphthalene	N/A	N/A	0/0	N/A	ND	ND	0/7	N/A
acenaphthene	160	160	1/1	SD19	ND	ND	0/7	N/A
dibenzofuran	96	96	1/1	SD19	ND	ND	0/7	N/A
fluorene	160	160	1/1	SD19	ND	ND	0/7	N/A
phenanthrene	60	1300	5/5	SD19	ND	460	4/8	SD12
anthracene	120	220	2/2	SD19	ND	ND	0/7	N/A

Table 4-1								
SUMMARY OF ORGANIC DATA FOR SEDIMENT SAMPLES								
Organic Parameter	St. Joseph River Background Sample Numbers SD17-SD22			St. Joseph River Sample Numbers SD09-SD16				
	Results Range (µg/kg)	Number of Detections ^a	Location of Maximum Concentration	Results Range (µg/kg)	Number of Detections ^a	Location of Maximum Concentration		
Semi-volatile Organic Compounds (Cont.)								
carbazole	190	190	1/1	SD19	ND	ND	0/8	N/A
fluoranthene	130	1,400	5/5	SD19	ND	980	5/8	SD12
pyrene	130	1,200	5/5	SD19	ND	1,400	5/8	SD12
butylbenzylphthalate	N/A	R	0/0	N/A	ND	ND	0/8	N/A
benzo(a)anthracene	64	690	5/5	SD20	ND	650	5/8	SD12
chrysene	69	630	5/5	SD20	ND	710	5/9	SD12
bis(2-ethylhexyl)phthalate	430	430	1/1	SD20	ND	1,400	5/8	SD12
di-n-octylphthalate	150	150	1/1	SD22	ND	ND	0/8	N/A
benzo(b)fluoranthene	71	730	5/5	SD20	ND	1,100	5/8	SD12
benzo(k)fluoranthene	46	460	4/4	SD20	ND	370	3/8	SD14
benzo(a)pyrene	160	720	4/4	SD20	ND	430	2/8	SD14
indeno(1,2,3-cd)pyrene	61	300	3/3	SD20	ND	130	1/8	SD10
benzo(g,h,i)perylene	420	420	1/1	SD20	ND	ND	0/8	N/A
Pesticides/PCBs								
Aldrin	ND	ND	0/6	N/A	ND	ND	0/8	N/A
Endosulfan I	ND	ND	0/6	N/A	ND	ND	0/8	N/A

Table 4-1

SUMMARY OF ORGANIC DATA FOR SEDIMENT SAMPLES

Organic Parameter	St. Joseph River Background Sample Numbers SD17-SD23				St. Joseph River Sample Numbers SD09-SD16			
	Results Range ($\mu\text{g/kg}$)	Number of Detections*	Location of Maximum Concentration		Results Range ($\mu\text{g/kg}$)	Number of Detections*	Location of Maximum Concentration	
Pesticides/PCBs (Cont.)								
Dieldrin	ND	ND	0/6	N/A	ND	ND	0/6	N/A
4,4'-DDE	ND	ND	0/6	N/A	ND	6.5	1/6	SD14
Endrin	ND	ND	0/6	N/A	ND	6.5	1/6	SD14
Endosulfan II	ND	ND	0/6	N/A	ND	ND	0/6	N/A
4,4'-DDD	ND	ND	0/6	N/A	ND	ND	0/6	N/A
4,4'-DDT	ND	ND	0/6	N/A	ND	ND	0/6	N/A
Methoxychlor	ND	ND	0/6	N/A	ND	ND	0/6	N/A
Endrin aldehyde	ND	ND	0/6	N/A	ND	ND	0/6	N/A
alpha chlordane	ND	0.75	1/6	SD20	ND	0.65	1/6	SD09
gamma chlordane	ND	1.2	2/6	SD22	ND	2.4	2/6	SD16
Aroclor-1254	ND	62	1/6	SD20	ND	190	4/6	SD14

Key:

ND: Compound was analyzed for but not detected.

N/A: Not applicable; maximum concentration corresponds to a non-detect concentration.

* Some results were rejected due to major violation of technical or QC protocols.

Source: Ecology and Environment, Inc. 1994.

Table 4-2

SUMMARY OF INORGANIC DATA FOR SEDIMENT SAMPLES

Analyte	Range Bay Background Sample Numbers SD01-SD03				Range Bay Sample Numbers SD04-SD08				Ponds Sample Numbers SD23-SD25			
	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration
Aluminum	1,360	3,000	3/3	SD02	1,530	13,400	5/5	SD04	1,740	10,600	3/3	SD25
Antimony	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	26.9	1/3	SD25
Arsenic	0.46	1.6	3/3	SD02	0.37	14.9	5/5	SD08	1.1	11.1	3/3	SD25
Barium	5.8	13.8	3/3	SD02	6.1	182	5/5	SD08	10.5	2,420	3/3	SD25
Beryllium	ND	ND	0/3	N/A	ND	0.8	2/5	SD05	ND	0.66	1/3	SD25
Cadmium	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	18.6	1/3	SD25
Calcium	16,100	24,300	3/3	SD01	1,080	85,300	5/5	SD08	35,000	85,300	3/3	SD24
Chromium	4.6	6.6	3/3	SD02	2.8	34.9	5/5	SD08	4.3	108	3/3	SD25
Cobalt	1.4	3.2	3/3	SD02	2.4	14.7	5/5	SD08	2.7	6.9	3/3	SD25
Copper	2.4	3.7	3/3	SD02	3.5	51.4	5/5	SD08	5.7	994	3/3	SD25
Iron	3,730	5,930	3/3	SD02	2,760	32,600	5/5	SD08	4,060	19,700	3/3	SD25
Lead	1.5	14	3/3	SD02	1.7	35.7	5/5	SD08	3.6	650	3/3	SD25
Magnesium	5,450	6,910	3/3	SD03	557	15,200	5/5	SD05	8,550	12,900	3/3	SD25
Manganese	102	114	3/3	SD02	71.2	1,860	5/5	SD08	148	822	3/3	SD25
Mercury	ND	ND	0/3	N/A	ND	0.3	1/8	SD04	ND	0.49	1/3	SD25
Nickel	4.2	7.9	3/3	SD02	4.3	30.2	5/5	SD08	6.2	216	3/3	SD25
Potassium	ND	337	1/3	SD02	ND	1,810	3/5	SD05	ND	ND	0/3	N/A

Table 4-3

SUMMARY OF INORGANIC DATA FOR SEDIMENT SAMPLES

Analyte	Range Bay Background Sample Numbers SD01-SD03				Range Bay Sample Numbers SD04-SD06				Ponds Sample Numbers SD23-SD25			
	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration
Silver	ND	ND	0/3	N/A	ND	ND	0/3	N/A	ND	1.5	1/3	SD25
Sodium	48.8	72.1	3/3	SD02	63.1	247	3/3	SD06	84.8	279	3/3	SD25
Vanadium	5.7	10.7	3/3	SD02	4.9	30.5	3/3	SD06	8.7	39	3/3	SD25
Zinc	10.4	18.3	3/3	SD02	10.1	194	3/3	SD06	21.6	5,250	3/3	SD25

4-28

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Table 4-2

SUMMARY OF INORGANIC DATA FOR SEDIMENT SAMPLES

Analyte	St. Joseph River Background Sample Numbers SD17-SD22			St. Joseph River Sample Numbers SD09-SD16				
	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration
Aluminum	1,610	10,500	6/6	SD17	2,090	11,900	8/8	SD12
Antimony	ND	ND	0/6	N/A	ND	ND	0/8	N/A
Arsenic	2.8	10.5	6/6	SD17	2	20.5	8/8	SD12
Barium	11.6	142	6/6	SD17	10	182	8/8	SD12
Beryllium	ND	ND	0/6	N/A	ND	1.3	3/8	SD10
Cadmium	ND	ND	0/6	N/A	ND	4.5	1/8	SD12
Calcium	4,270	77,100	6/6	SD18	1,910	67,800	8/8	SD10
Chromium	6.1	33.5	6/6	SD17	7.7	38.9	8/8	SD10
Cobalt	1.9	7	6/6	SD17	2	13.7	8/8	SD12
Copper	3.3	44.3	6/6	SD17	2.7	60.7	8/8	SD10
Iron	5,080	31,200	6/6	SD20	4,720	33,000	8/8	SD12
Lead	5.4	57.9	6/6	SD17	1.7	84	8/8	SD14
Magnesium	1,190	21,500	6/6	SD18	1,080	11,200	8/8	SD12
Manganese	81	1,470	6/6	SD17	86.6	2,330	8/8	SD12
Mercury	ND	2.8	1/6	SD17	ND	0.27	1/8	SD13
Nickel	5.5	22.7	6/6	SD17	4.5	26.5	8/8	SD12
Potassium	ND	9050	1/6	SD17	ND	ND	0/8	N/A

SUMMARY OF INORGANIC DATA FOR SEDIMENT SAMPLES

Analyte	St. Joseph River Background Sample Numbers SD17-SD23				St. Joseph River Sample Numbers SD09-SD16			
	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration	Results Range (mg/kg)		Number of Detections	Location of Maximum Concentration
Silver	ND	ND	0/6	N/A	ND	ND	0/6	N/A
Sodium	57.5	189	6/6	SD17	64.1	348	8/8	SD14
Vanadium	6.3	230	6/6	SD17	7.2	28.3	8/8	SD14
Zinc	14.3	154	6/6	SD17	11.3	198	8/8	SD12

ND: Analyte was analyzed for but not detected.
N/A: Not applicable; maximum concentration corresponds to a non-detect result.

Source: Ecology and Environment, Inc. 1994.

Table 4-3

SUMMARY OF ORGANIC DATA FOR SURFACE WATER SAMPLES

Organic Parameter	Range Bay Background Sample Numbers SW01-SW03			Range Bay Sample Numbers SW04-SW06			Ponds Sample Numbers SW23-SW25					
	Results Range (µg/L)	Number of Detections	Location of Maximum Concentration	Results Range (µg/L)	Number of Detections	Location of Maximum Concentration	Results Range (µg/L)	Number of Detections	Location of Maximum Concentration			
Volatile Compounds												
trichloroethene	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
Semivolatile Compounds												
diethylphthalate	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
di-n-butylphthalate	ND	0.7	2/3	SW02	ND	ND	0/5	N/A	ND	ND	0/3	N/A
bis(2-ethylhexyl)phthalate	ND	0.7	2/3	SW01/03	ND	1.0	3/5	SW06	ND	ND	0/3	N/A
di-n-octylphthalate	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	ND	0/3	N/A
Pesticides/PCBs												
Endrin aldehyde	ND	0.014	1/3	SW03	ND	ND	0/5	N/A	ND	ND	0/3	N/A

Table 4-3
SUMMARY OF ORGANIC DATA FOR SURFACE WATER SAMPLES

Organic Parameter	St. Joseph River Background Sample Numbers SW17-SW22				St. Joseph River Background Sample Numbers SW09-SW16			
	Result Range (µg/L)	Number of Detections	Location of Maximum Concentration		Result Range (µg/L)	Number of Detections	Location of Maximum Concentration	
Volatile Compounds								
trichloroethene	ND	ND	0/6	N/A	ND	3/5	1/8	SW15
Semivolatile Compounds								
diethylphthalate	ND	1.0	1/6	SW19	ND	3	1/8	SW13
di-n-butylphthalate	ND	0.6	2/6	SW20	ND	ND	0/8	N/A
bis(2-ethylhexyl)phthalate	ND	1.0	4/6	SW19	ND	0.8	2/8	SW09
di-n-octylphthalate	ND	6.0	6/6	SW19	ND	4	3/8	SW16
Pesticides/PCBs								
Endrin aldehyde	ND	0.03	3/6	SW22	ND	ND	0/7	N/A

Key:

ND: Analyte was analyzed for but not detected.

N/A: Not applicable; maximum concentration corresponds to a non-detect result.

Source: Ecology and Environment, Inc. 1994.

Table 4-4

SUMMARY OF INORGANIC DATA FOR SURFACE WATER SAMPLES

Analyte	Range Bay Background Sample Numbers SW01-SW03				Range Bay Sample Numbers SW04-SW06				Ponds Sample Numbers SW23-SW25			
	Results Range (µg/L)		Number of Detections	Location of Maximum Concentration	Results Range (µg/L)		Number of Detections	Location of Maximum Concentration	Results Range (µg/L)		Number of Detections	Location of Maximum Concentration
Aluminum	226	333	3/3	SW01	97.8	4,170	5/5	SW05	ND	304	2/3	SW25
Antimony	ND	ND	0/3	N/A	ND	93.6	2/5	SW05	ND	ND	0/3	N/A
Arsenic	ND	1.3	1/3	SW01	ND	2	3/5	SW05	ND	ND	0/3	N/A
Barium	53.1	53.1	3/3	SW01/02/03	52.8	76.3	5/5	SW05	13.3	106	3/3	SW25
Calcium	83,200	84,000	3/3	SW02	78,400	92,500	5/5	SW05	48,500	59,200	3/3	SW24
Copper	ND	ND	0/3	N/A	ND	ND	0/5	N/A	ND	19.9	1/3	SW25
Iron	329	368	3/3	SW01	300	4,220	5/5	SW05	18.8	651	3/3	SW25
Lead	ND	1.1	1/3	SW01	ND	2.8	1/5	SW05	ND	41.1	1/3	SW25
Magnesium	25,900	26,300	3/3	SW02	20,900	28,600	5/5	SW05	14,800	16,100	3/3	SW23
Manganese	29.9	31.4	3/3	SW03	28	139	5/5	SW05	18.2	286	3/3	SW24
Potassium	3,580	3,920	3/3	SW01	1,880	5,300	5/5	SW05	1,360	2,210	3/3	SW23
Selenium	ND	1.5	1/3	SW01	ND	ND	0/5	N/A	ND	ND	0/3	N/A
Sodium	14,300	14,800	3/3	SW02	9,900	15,600	5/5	SW05	4,940	6,190	3/3	SW23
Zinc	ND	13.8	1/3	SW01	12.9	50.8	5/5	SW04	ND	155	2/3	SW25

Table 4-4

SUMMARY OF INORGANIC DATA FOR SURFACE WATER

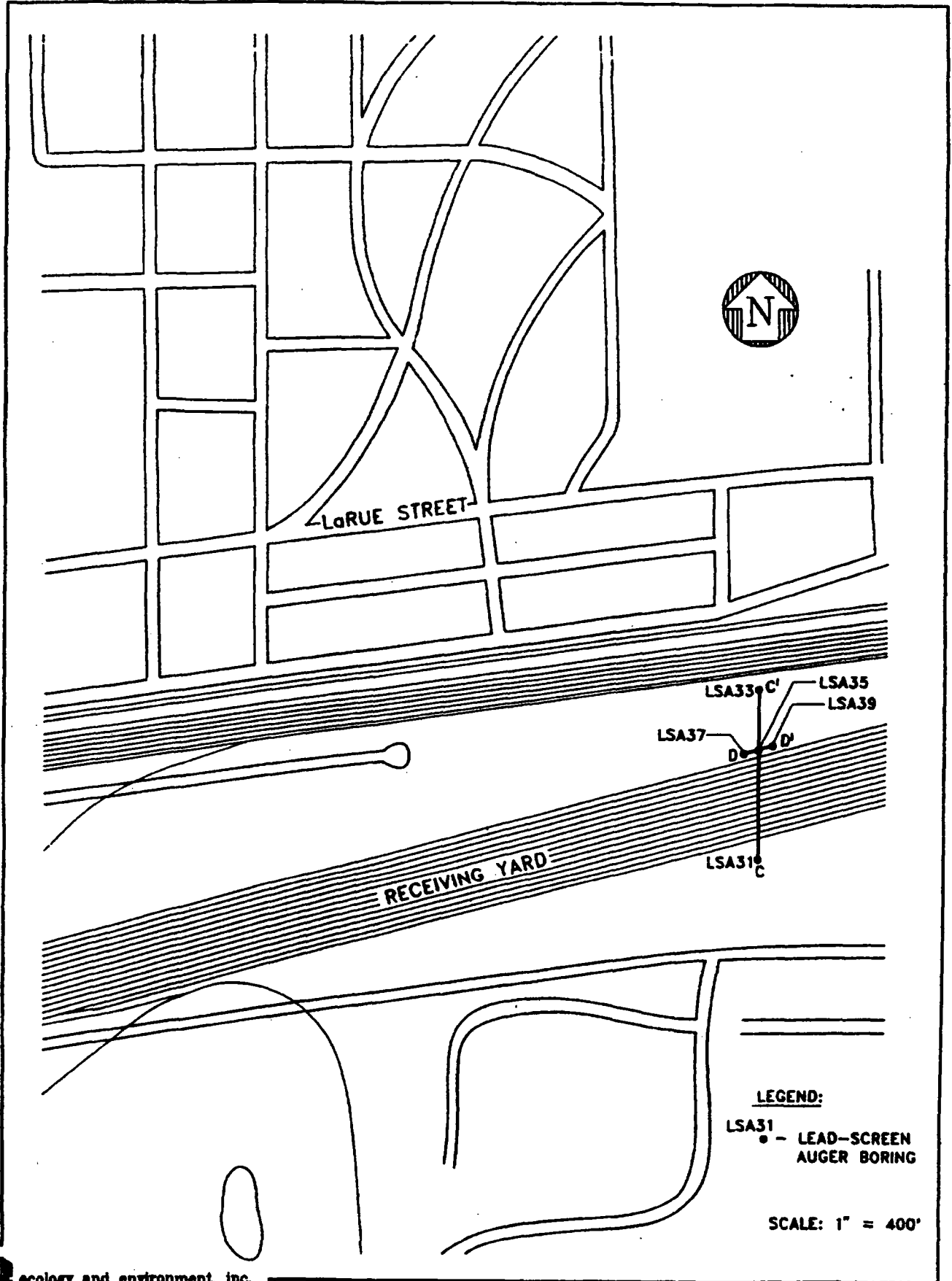
Analyte	St. Joseph River Background Sample Numbers SW17-SW22				St. Joseph River Sample Numbers SW09-SW16			
	Results Range (µg/L)		Number of Detections	Location of Maximum Concentration	Results Range (µg/L)		Number of Detections	Location of Maximum Concentration
Aluminum	398	572	6/6	SW22	152	2,500	8/8	SW16
Arsenic	ND	70.9	1/6	SW22	ND	72.1	5/8	SW16
Arsenic	1.1	1.2	3/6	SW20/21	ND	2.6	3/8	SW16
Barium	51.8	55.3	6/6	SW17	50.2	79.3	8/8	SW16
Calcium	73,100	75,000	6/6	SW21	76,900	85,300	8/8	SW16
Copper	ND	ND	0/6	N/A	ND	ND	0/8	N/A
Iron	513	621	6/6	SW17	352	3,600	8/8	SW16
Lead	ND	1.19	1/6	SW22	ND	5.2	2/8	SW16
Magnesium	19,600	20,100	6/6	SW21	20,600	21,800	8/8	SW12/16
Manganese	36.4	41,470	6/6	SW18	28	300	8/8	SW16
Potassium	2,340	2,910	6/6	SW19	1,820	2,910	8/8	SW16
Selenium	ND	ND	0/6	N/A	ND	ND	0/8	N/A
Sodium	8,810	10,100	6/6	SW21	9,370	10,200	8/8	SW09
Zinc	ND	ND	0/6	N/A	ND	20.7	2/8	SW09

Key:

ND: Analyte was analyzed for but not detected.

N/A: Not applicable; maximum concentration corresponds to a non-detect result.

Source: Ecology and Environment, Inc. 1994.

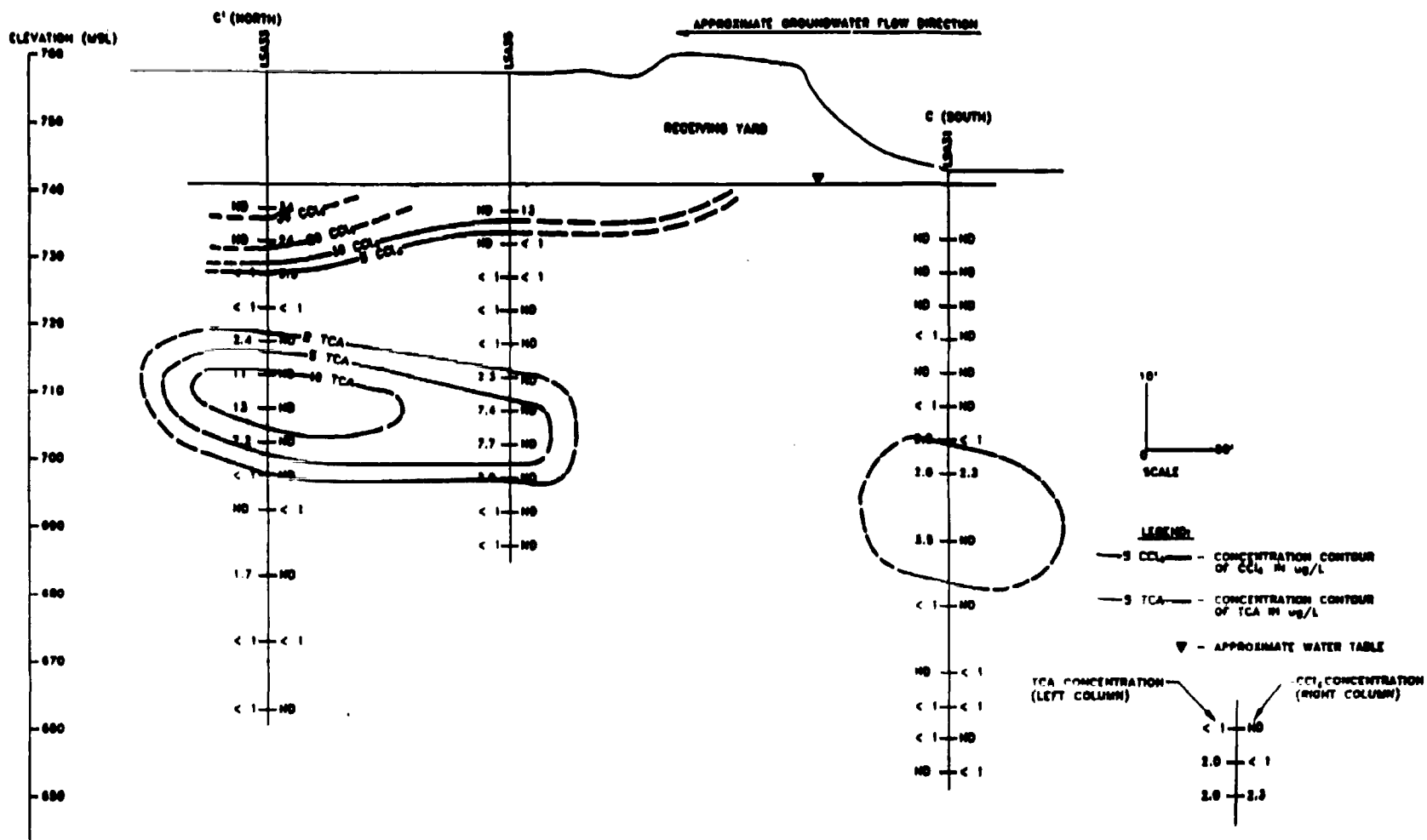


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FIGURE 4-1
 CROSS-SECTION LOCATIONS - C-C' AND D-D'

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FIGURE 4-2 CROSS SECTION C-C'
CCl₄ AND TCA
CONCENTRATIONS IN ug/L

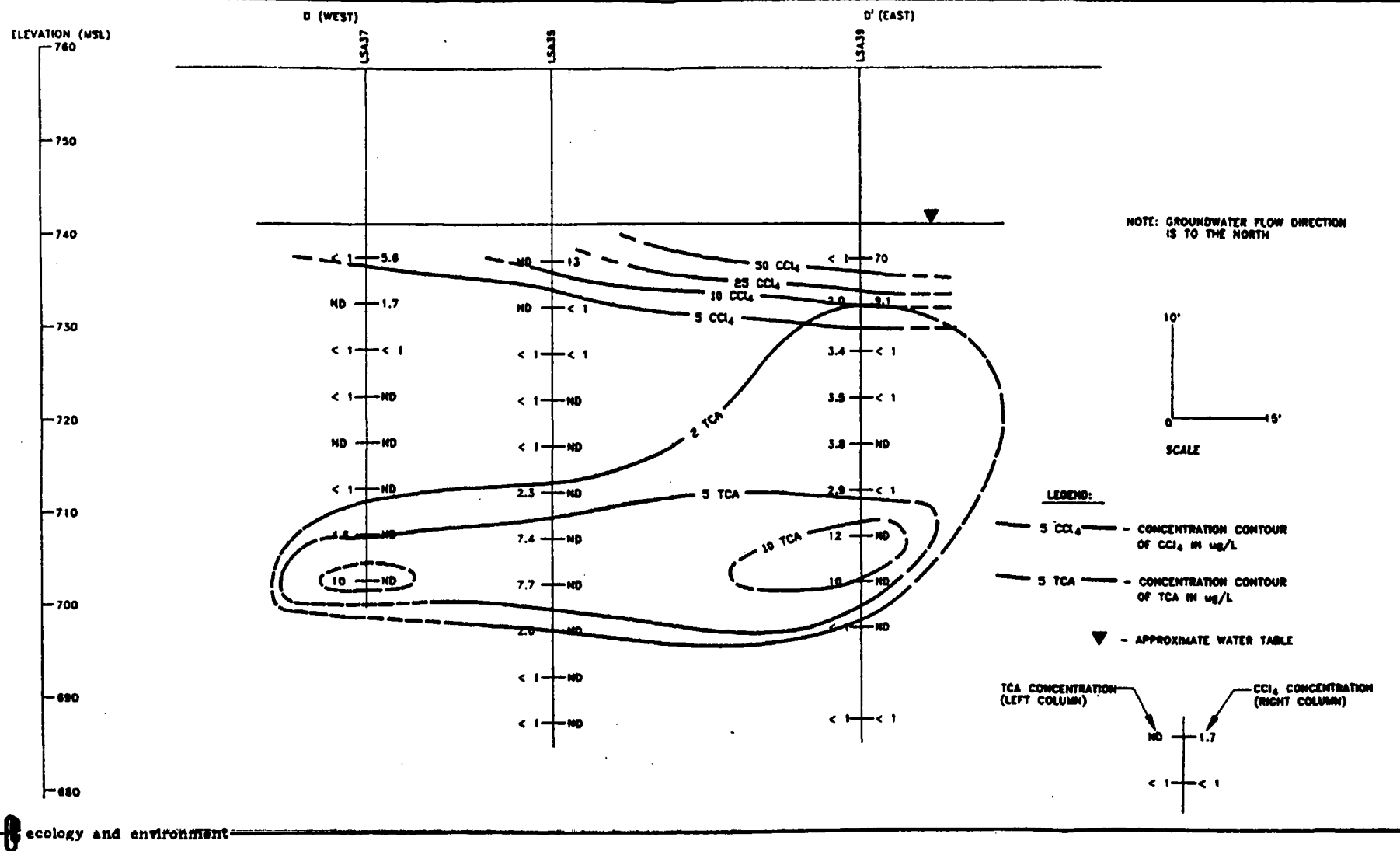
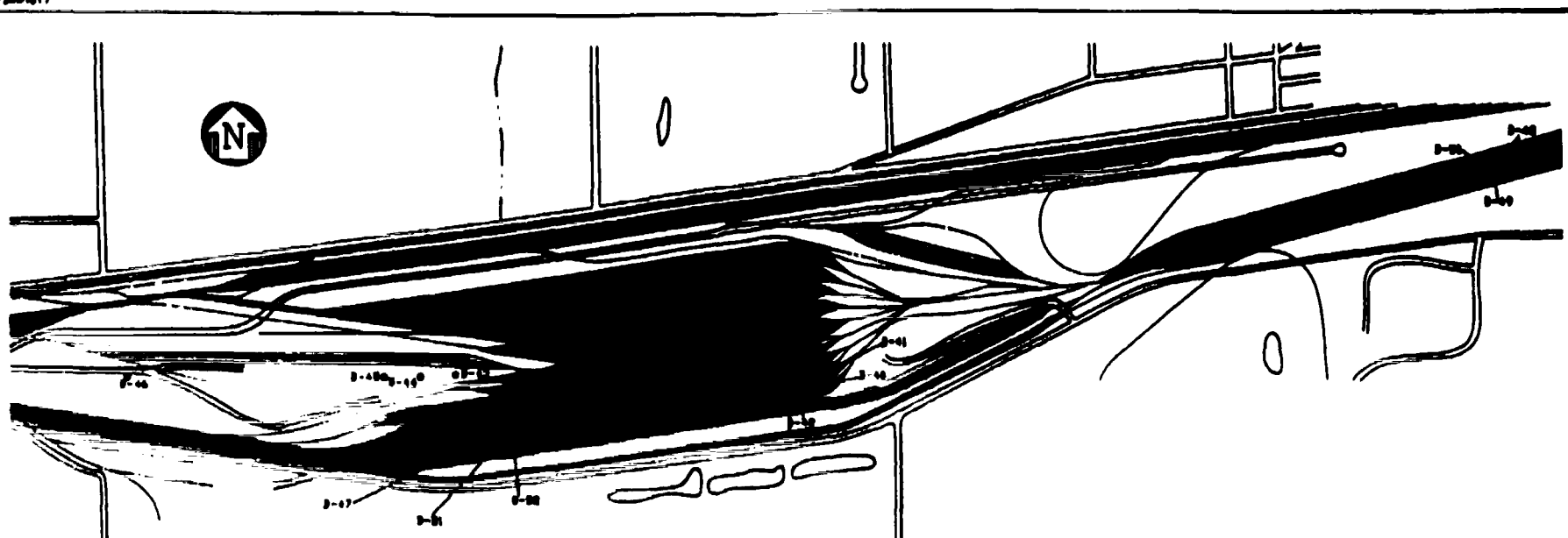


FIGURE 4-3 CROSS SECTION D-D'
 CCl_4 AND TCA
 CONCENTRATIONS IN $\mu\text{g/L}$

4-37



BORING NUMBER SAMPLE INTERVAL (IN FEET BELOW GROUND SURFACE) CARBON TETRACHLORIDE (IN ug/kg s) TRICHLOROETHYLENE (IN ug/kg s) COMMENTS	TRACE 00/00 AREA										SPILLED OIL AREA											
	0-00										0-01		0-02			0-03			0-04			
	0-10	10-20	20-34	30-50	110-120	110-120	130-131	140-142	144-145	100-100	0-7	71-23	13-20	7-8	71-23	20-20	7-8	13-13	23-20	7-8	13-17	23-20
	ND	1100	64,000	ND	ND	10	ND	ND	ND	ND	ND	71,000	1,300	ND	20,000	10	ND	ND	ND	ND	ND	ND
	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0	ND
	NONE										NONE		NONE			NONE			NONE			

BORING NUMBER SAMPLE INTERVAL (IN FEET BELOW GROUND SURFACE) CARBON TETRACHLORIDE (IN ug/kg s) TRICHLOROETHYLENE (IN ug/kg s) COMMENTS	SPILLED OIL AREA			BURIED TANK 000			TRACE 00/00 AREA				RECEIVING TANK				TRACE 00/00 AREA							
	0-00			0-00			0-07				0-00		0-00		0-00		0-01		0-00			
	7-8	10-17	23-20	7-8	17-10	10-21	0-2	0-4	10-14	ND	10-10	10-20	0-8	10-10	20-24	0-2	0-2	12-14	0-2	0-4	0-8	0.0-1.0
	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	21	ND	ND	20	20	20	ND	ND	ND	ND	ND	ND
	ND	ND	17	ND	ND	ND	ND	70	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	20	20	ND
COMMENTS	NONE			7-8 SAMPLE ANALYZED USING REMOTE MONITORING DETECTION LIMITS			NONE				NONE		NONE		NONE		NONE		-1.1 mg/kg (0.0-0.5 mg/kg) (TOTAL) (0-0)			

• = SOIL BORING SAMPLES NOT SENT FOR CLP ANALYSIS ARE NOT LISTED IN THIS TABLE

SCALE
1" = 700'

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FIGURE 4-4

PHASE II
SOIL BORING LOCATIONS
AND SELECTED ORGANIC
ANALYTICAL RESULTS

4-80

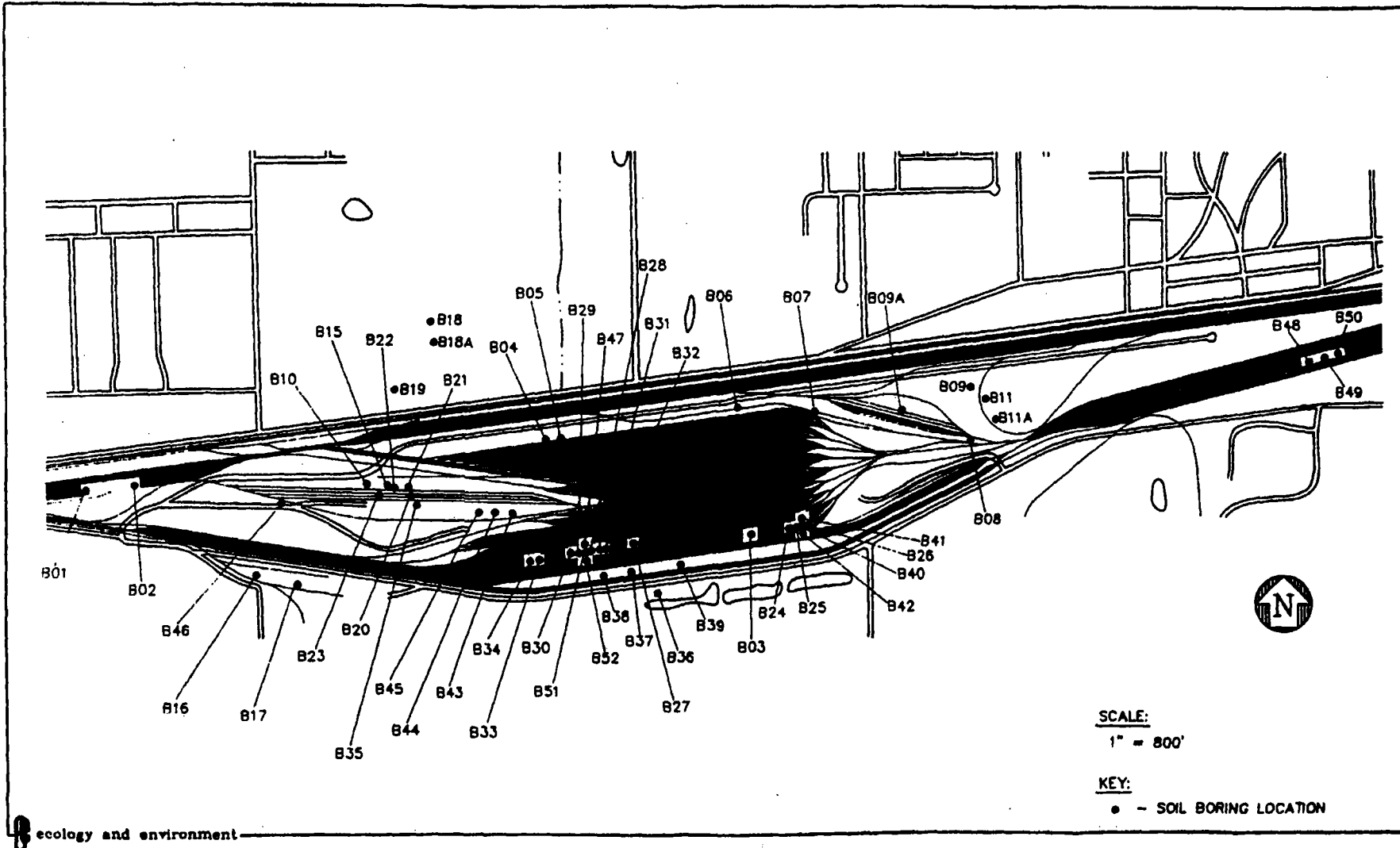


FIGURE 4-5

PHASE I, II and III
SOIL BORING
LOCATIONS MAP

The chemicals of potential concern at the Conrail Site include the chlorinated aliphatic hydrocarbons, CCl_4 and TCE. This section discusses the fate and transport of these chemicals and the other chlorinated aliphatic hydrocarbons most frequently detected on site. The chemicals of potential concern discussed in the HHE portion of the risk assessment include the major chemicals discussed in this section in addition to other chemicals detected on site that may pose a risk to human populations (see Section 6).

5.1 GENERAL FACTORS INFLUENCING TRANSPORT AND FATE AT THE SITE

0126

spill event has occurred. The liquids released to the site may have included single chemicals and mixtures of chemicals. The chemical composition of the liquids released and soil composition will affect the mobility and fate of the chemicals of potential concern contained within the liquids.

Table 5-1 presents a list of chlorinated aliphatic hydrocarbons of potential concern and their associated chemical properties. A discussion of the general physical and chemical properties affecting the mobility of the major chlorinated aliphatic hydrocarbons present on the Conrail Site, and possible transformation reactions follows.

5.1.1 Volatilization

Vapor pressure is a relative measure of the volatility of a chemical in its pure state, and represents a compound's tendency to evaporate. It is an important determinant of the rate of vaporization from waste sites. A compound's volatilization rate from water depends on its vapor pressure and water solubility. Volatilization is an important factor when organic contamination is within or close to the unsaturated zone.

The Henry's Law constant is the ratio of the compound's vapor pressure (in atmospheres) to its solubility in water (in moles/m³). As the concentration of the solute approaches zero, Henry's Law becomes a more accurate measure than vapor pressure for predicting volatilization of the chemical to air from water. Compounds with Henry's Law constants greater than approximately 10⁻³ can be expected to volatilize readily from water. Those with values ranging from 10⁻³ to 10⁻⁵ volatilize less readily, while compounds with values less than 10⁻⁵ volatilize slowly (Lyman, Reehl, and Rosenblatt 1982). Values for Henry's Law constant (H) are defined by the following equation, where MW is the mole weight (see Table 5-1):

$$H \text{ (atm} \cdot \text{m}^3\text{)/(mole)} = \frac{\text{vapor pressure (atm)} \times \text{MW (g/mole)}}{\text{water solubility (g/m}^3\text{)}} \quad [\text{Eq. 5-1}]$$

The chlorinated aliphatic hydrocarbons detected at the Conrail Site are volatile compounds with vapor pressures ranging from 18.5 to 349 mm Hg (for tetrachloroethane

[PCE] and methylene chloride, respectively) and with Henry's Law constants ranging from 2.03×10^{-3} to 2.59×10^{-2} atm-m³/mol (for methylene chloride and PCE, respectively) (see Table 5-1). Some of the Henry's Law constants shown in Table 5-1 were compiled from references that are independent of the sources used to compile the solubility and vapor pressure values. Use of Equation 5-1 to derive Henry's Law constants will result in small variations from the values shown. This is a reflection of numerous components that contribute to the total uncertainty associated with the values listed in this table. The Henry's Law values shown in Table 5-1 suggest that the major chemicals of potential concern at the Conrail Site are expected to volatilize readily from water. Volatilization removes contaminant mass from groundwater, resulting in the transfer of significant quantities of mass from groundwater to soil gas. Volatilization of dissolved organic solutes from groundwater that is in contact with a gas phase can be modeled using equilibrium concepts based on Henry's Law. This mass law equation relates the concentration of dissolved chemical in solution to the partial pressure of the same chemical in a gas phase (i.e., air or soil gas) in contact with the solution. The process is controlled by the concentration of the organic solute in the vapor and water. An estimate of the total amount of TCE and CCl₄ that has undergone volatilization has not been calculated. However, volatilization is a migration pathway that is addressed in detail in the HHE (Section 6).

• 5.1.2 Liquid Transport

In pure form the chemicals of potential concern at the Conrail Site would exist as liquids over the range of ambient air pressures and temperatures expected to occur at the site. The liquid densities range from 1.28 for 1,2-dichloroethene (1,2-DCE) to 1.63 g/mL for PCE (see Table 5-1). These densities are greater than the density of groundwater (approximately 1 g/mL). Spills of these hydrocarbons that reach the groundwater would have the potential to migrate downward through the groundwater column.

Water solubility (the maximum concentration of a chemical that dissolves in pure water at a specific temperature and pH) is a critical property affecting the environmental transport of a chemical. Highly water soluble chemicals can be rapidly leached from soils or waste and are generally mobile in groundwater, traveling through advection. Advection is the process by which the bulk motion of flowing groundwater transports chemicals. The

movement of chemicals in the water is also influenced by dispersion. Dispersion is the result of molecular diffusion (movement from high to low concentrations) and mechanical mixing (the result of variations in groundwater flow). Dispersion spreads the contaminant mass in three dimensions beyond the space it normally would occupy due to advection alone. The chlorinated aliphatic hydrocarbons detected at the Conrail site are moderately to poorly water-soluble, ranging from 8×10^2 mg/L for CCl_4 to 2×10^4 mg/L for methylene chloride (see Table 5-1).

The groundwater contaminant distribution suggests that chemical releases likely consisted of single-component liquid DNAPLs. DNAPL chemicals are immiscible with and denser than water. Their immiscibility and high density enable DNAPL constituents released to a porous medium to penetrate the unsaturated zone and migrate downward into the saturated zone as a separate nonaqueous phase. DNAPLs can overcome vapor pressure in the unsaturated zone and pore pressure in the saturated zone if conditions are favorable (i.e., large releases of DNAPL), and flow downward until their progress is finally impeded. Impediments to DNAPL flow could be a low-permeability geologic unit, a cohesive, continuous clay unit, or a dense shale bedrock unit.

A characteristic of DNAPLs that also influences vertical migration is their relatively low viscosities in comparison to water. It has been shown in experimental models by Schuille (1988) that a DNAPL of lower viscosity than water will sink more quickly than water through the unsaturated zone. Upon encountering a zone of lower permeability or a capillary fringe, however, the DNAPL migration rate is substantially slowed and will only pass through these layers at high hydraulic conductivity pathways, whereas the rate of groundwater migration is not affected to the same degree. A DNAPL will descend faster in the unsaturated zone than water because of its density and viscosity. In the saturated zone, however, the pressures that must be overcome and the effects of buoyancy prevent rapid movement of the DNAPL.

During the DNAPL spreading process through the unsaturated zone, the DNAPL will begin to occupy available pore space of the soil. A DNAPL will progress more rapidly through a moist unsaturated zone than a dry unsaturated zone because more pore spaces are wetted in a moist medium and thus are not available to retain the DNAPL. Besides moisture, the retention capacity of a soil can be influenced by low-permeability layers, capillary fringes,

and depressions in the low-permeability layers, all of which can cause the released DNAPL to concentrate. The residual saturation of a DNAPL in the unsaturated zone is typically present as immobile, semi-isolated masses and stringers. Finer-grained soil will typically exhibit a greater amount of residual saturation.

If the volume of DNAPL is sufficiently large and exceeds the retention capacity of the unsaturated zone, the DNAPL can flow into the saturated zone. In this case, the DNAPL must have sufficient driving fluid pressure force to overcome the pore pressure existing between the groundwater and the porous media in order to penetrate the saturated zone. Only then can the DNAPL progress downward through the saturated zone, displacing and replacing the groundwater in its path. The residual saturation will exist as droplets of DNAPL within the porous, saturated media. Residual saturation values are higher for the saturated zone than for the unsaturated zone—reaching a maximum value in porous media with a hydraulic conductivity of 1×10^{-4} m/sec (Schwille 1988). DNAPLs will have great difficulty penetrating strata with hydraulic conductivities of less than approximately 1×10^{-4} m/sec, even with a continuous source of DNAPL. The continuity and orientation of such inhibiting layers have a direct effect on the distribution of the DNAPL in the subsurface such that pooled or residual DNAPLs in the subsurface may be located at a considerable lateral distance from the release location on the surface. If the DNAPL progresses to the bottom of the aquifer, and the initial release is large enough to exceed the residual saturation values of the unsaturated and saturated zones, the DNAPL will pool at the bottom, essentially impervious layer, and spread laterally to the extent the excess DNAPL volume allows.

The residual DNAPL in the unsaturated and saturated zones and the pooled DNAPL at the bottom of the aquifer are all aspects of a contributing source to groundwater contamination. It is this extended source that will have significant implications on the groundwater contamination pattern. Although DNAPLs are not miscible with water, contaminants from the DNAPL sources are drawn into solution by infiltrating aquifer recharge and groundwater flow; these contaminants, however, are seldom if ever detected close to their respective solubility concentrations. Groundwater flow patterns will not affect the initial descent of the DNAPL; however, they can influence contaminant concentrations in solution.

It has been estimated that the fraction of total pore space containing DNAPL typically ranges from 10% to 20% in unsaturated soils and 15% to 50% in saturated soils through

which a DNAPL has traveled (Mercer and Cohen 1990). This indicates that a sizable volume may remain in the area of the initial DNAPL source. In the groundwater, soluble portions of the residual DNAPL will be leached from these areas by infiltrating water or moving groundwater and travel by advection. Chemicals that compose DNAPLs are only rarely found in groundwater at their solubility limits because of diffusional limitations and dispersion. Concentrations as low as 1 % of the chemical solubility may indicate the presence of a DNAPL (EPA 1992a).

5.1.3 Sorption and Retardation

Sorption of aliphatic chlorinated hydrocarbons is primarily dependent on the fraction of soil organic matter in the aquifer. In addition to organic matter, residual petroleum oils may also serve as a sorptive medium for organic chemicals (Boyd and Sun 1990).

The organic carbon partition coefficient (K_{oc}) is a chemical-specific measure of the tendency for organics to be sorbed by the organic matter found in soil and sediment and is expressed as:

$$K_{oc} = \frac{\text{mg of solute sorbed on organic carbon/kg organic carbon}}{\text{mg of solute/L of solution}} \quad [\text{Eq. 5-2}]$$

K_{oc} values for the compounds listed in Table 5-1 range from 8.8 to 152 with higher values indicating greater sorption potential. Generally, it is expected that sorption will be an important transport process affecting the subsurface migration of the chemicals of potential concern on site.

Sorption is often described in terms of the distribution of the chemical between the groundwater and aquifer materials. The distribution or partition coefficient (K_d) of chemicals having solubilities no more than approximately 0.001 mol/L (Domenico and Schwartz 1990) is often described using a linear Freundlich isotherm (Karickhoff, Brown, and Scott 1979) where K_d is defined as:

$$K_d = \frac{\text{mg of solute sorbed on the solid phase/kg of solid phase}}{\text{mg of solute/L of solution}} \quad [\text{Eq. 5-3}]$$

Values of K_d will vary for each chemical with the composition of the sorbing media. Since the organic carbon content of the aquifer has a significant impact on sorption, K_d is commonly normalized for the amount of organic carbon in the soil (Karickhoff, Brown, and Scott 1979) such that:

$$K_d = K_{oc} f_{oc}, \text{ where } f_{oc} = \text{the weight fraction of organic carbon in the soil.}$$

The fraction of organic carbon was measured in four samples collected from the subsurface unconsolidated materials, ranging in depth from 7.5 to 24.5 feet BGS, during the Phase II investigation (see Appendix B for analytical data) (E & E 1991c). Sample results indicated that the carbon content of the subsurface unconsolidated materials varied from 0.30% to 2.38%, or an f_{oc} of 0.0030 to 0.0238, respectively. Using K_{oc} values given in Table 5-1 and the measured minimum and maximum range values for f_{oc} , calculated K_d values were the expected low values typically associated with glacial material and ranged from 0.026 to 3.6 mL/g for methylene chloride and TCA, respectively.

When no solute is sorbed to the solid phase, K_d is equal to zero, and it would be assumed that the chemical would move at the same speed as the groundwater. K_d is used to determine the retardation (R_f) of a chemical with a linear equilibrium partitioning in the groundwater, where:

$$R_f = \frac{\text{time for retarded chemical to reach a given point}}{\text{time for groundwater (or nonsorbing chemical) to reach a given point}} \quad [\text{Eq. 5-4}]$$

R_f can be measured as:

$$R_f = 1 + \frac{\rho_b K_d}{n} \quad [\text{Eq. 5-5}]$$

Where $\rho_b = \rho_s (1 - n)$ = bulk density and n is total porosity. The total porosity is assumed to be 0.4 for the unconsolidated glacial outwash material (EPA 1983). This value is consistent with the value chosen for effective porosity (0.25) in Section 3 (EPA 1983).

A range of calculated R_f factors is presented for each of the chemicals of potential concern on site in Table 5-2. These ranges were developed using low total porosity and high organic carbon content (these assumptions are conservative with respect to estimates of the mass removal rate through natural attenuation, i.e., a higher R_f factor, increasing the time required for the chemicals to leave the source areas) and using high total porosity and low organic carbon content (these assumptions are conservative with respect to the estimates of the time required to reach sensitive populations, i.e., a lower R_f factor, decreasing the time required for chemicals to leave the site). Values range from 1.1 (for methylene chloride) to 23.4 (for TCA), indicating that the aliphatic hydrocarbons present on site may be traveling at 90% (1/1.1) to 4% (1/23.4) of the rate of groundwater. Using a low organic carbon fraction (0.003) and moderate total porosity (0.40) assumed to be typical for the predominant groundwater flow paths in the unconfined aquifer at the site, retardation factors of approximately 2.5 were calculated for the two major chemicals of potential concern (CCl_4 and TCE) detected on site (see Table 5-2). The low organic carbon fraction was chosen because of the likelihood that it is representative of the sand and gravel aquifer material that composes the zones in the aquifer in which the majority of groundwater flow occurs. A retardation factor of 2.5 implies that contamination will travel at 40 percent of the rate of groundwater.

5.1.4 Transformation Reactions

Hydrolysis and oxidation reactions are generally not a major factor affecting chlorinated hydrocarbon contamination in groundwater systems. Chlorinated methanes (CCl_4 , CHL, methylene chloride), chlorinated ethenes (PCE, TCE, dichloroethenes [DCE], and vinyl chloride), and TCA undergo sequential reductive dehalogenation under anoxic conditions as reported by Smith and Dragun in 1984 (see Figure 5-1).

PCE in anaerobic soils has been shown to be slowly transformed through a series of biotic and abiotic reactions to TCE, 1,1-DCE, 1,2-DCE, and vinyl chloride (Bourwer and McCarty 1983, Vogel and McCarty 1985). Anaerobic biotransformation of TCA to 1,1-DCE and 1,1-dichloroethane has also been observed (Vogel, Griddle, and McCarty 1987). Vinyl chloride may accumulate as the final product of this process if other carbon sources are present (Vogel and McCarty 1985), but, in the absence of such sources, vinyl chloride may be further degraded and ultimately mineralized (Parsons, Wood, and DeMarco 1984; Kleopfer *et al.* 1985; Wilson *et al.* 1986). The rate and extent of transformation reactions are highly dependent on site-specific factors such as nutrient availability and microbial composition of the soil. In groundwater systems, biotransformation processes are generally limited to the upper 20 feet of the unconfined aquifer because that is where the highest concentrations of bacterial populations are usually found.

5.2 SITE-SPECIFIC FATE AND TRANSPORT MECHANISMS

The general processes and mechanisms discussed previously are applied in this subsection in order to present a site-specific analysis. The nature and extent of contamination in the study area and the chemical data are more fully discussed in Section 4.

5.2.1 Movement, Mass, and Loading of Chemicals of Potential Concern

Detailed below are estimates of chemical travel time in groundwater, contaminant mass remaining on site, loading to the river, and rate of natural attenuation.

5.2.1.1 Travel Time

Contamination has been detected in the homeowner wells since as early as 1986 (Roy F. Weston 1986b). Groundwater sampling in the Charles Avenue residential area was completed in 1987 for the Peerless-Midwest, Inc., pumping test. The concentrations observed during the pumping test (388 ppb CCl_4 and 2,495 ppb TCE) (Peerless [no date]) are not substantially different than the results obtained from the RI/FS investigation.

Groundwater data from monitoring wells on site supplemented by LSA data delineate plumes of CCl₄ and TCE emanating from the identified source areas and extending to the St. Joseph River, a distance of approximately 12,000 feet.

Since groundwater travels in the range of 10 to 2,000 feet/year at the site, it would take approximately 1,200 to 6 years, respectively, for groundwater from the source areas in the Conrail railyard to reach the St. Joseph River. If CCl_4 and TCE were not retarded, then it would travel with the groundwater through advection at the rate of groundwater flow.

Carbon matter with available sorptive capacity in the aquifer materials would allow sorption, resulting in slower travel times for chemicals dissolved in the groundwater compared with groundwater travel times. It is estimated that CCl_4 and TCE will travel at approximately 40 percent of the rate of groundwater in portions of the aquifer where organic carbon with sorptive capacity is available.

The estimated travel time for chemicals to migrate from the source areas to the St. Joseph River would range from six years for unretarded chemical transport through higher-conductivity portions of the aquifer to almost 3,000 years for retarded chemical transport through lower-conductivity portions of the aquifer.

5.2.1.2 Contaminant Mass

The site hydrogeology indicates that the groundwater in the unconsolidated aquifer materials follows a flow path with downward and horizontal components. Near the St. Joseph River an upward component of flow is observed in nested wells as groundwater discharges to the river. Groundwater samples indicate that all depth intervals of the unconsolidated aquifer have detected concentrations of chemicals. This suggests that in the source areas, residual DNAPL may extend the full length of the aquifer. Horizontal groundwater flow paths at all depths will have the potential of passing through residual contamination. The observed variability in groundwater contaminant distribution supports the hypothesis that a tortuous path of residual DNAPL extends to a significant depth in the unconsolidated aquifer materials.

The total mass (M_T) of chemical remaining in the aquifer as contaminants in the groundwater and sorbed to aquifer materials can be estimated as:

$$M_T = R_f C n V \quad [\text{Eq. 5-6}]$$

where R_f is the retardation factor, C is the concentration of the chemical in the groundwater, n is the total porosity, and V is the bulk aquifer volume (EPA 1989a). This relationship

applies because the concentration of the chemical in the groundwater is less than half of the solubility of the chemical. Any estimated mass of residual DNAPL would need to be added to the total mass of the chemical as estimated through the equation.

The current plume configuration has been delineated by analytical data obtained from groundwater samples collected from nested wells and LSA data. These data indicate a wide variation in chemical concentrations measured in groundwater. The range of variability and the extent of the CCl_4 and TCE plumes are similar. For the purposes of this estimate, it is assumed that although these two plumes are not congruent, the contaminant distribution and total contaminant mass within each plume is the same. The dimensions of each plume are approximately 3,000 feet wide, by 12,000 feet long, by 125 feet deep, or approximately $4.5 \times 10^9 \text{ ft}^3$ (1.274×10^{11} liters) in size.

The variability of the chemical concentrations within the plume over which the contaminant mass is being measured requires assigning concentrations to volumes of the plume to obtain a weighted average. A simplified estimate of the total chemical mass in the aquifer is obtained by summing the mass of chemicals in each volume, as shown in Table 5-3. For this simplified estimate, it is assumed that the weighted average concentration for CCl_4 and TCE is the same. The concentration for each volume fraction was approximated from LSA sampling and monitoring well sampling results. At 1.274×10^{11} liters in size, 80 percent of the total plume's volume contains contamination at $1 \mu\text{g/L}$ or less. The estimate in Table 5-3 shows that the majority of the total mass of chemical is contained in 0.11 percent of the total volume of the plume. The highest TCE concentration detected was $15,000 \mu\text{g/L}$. Although not reflected in the analytical results, it was assumed that the estimated sum of the mass of TCE includes 0.01 percent of the aquifer at $100,000 \mu\text{g/L}$ TCE. CCl_4 has been detected at this level, but TCE has not. This assumption for TCE is compensated by the fact that TCE is observed to occupy a larger volume than CCl_4 for the $1,000 \mu\text{g/L}$ to $10,000 \mu\text{g/L}$ concentration range. Based on the stated assumptions, this simplified calculation is justified to give a reasonable estimate of the dissolved and sorbed mass of CCl_4 and TCE.

The estimated total mass of chemicals in the groundwater and sorbed to aquifer materials is estimated at 4,557,000 grams or 10,000 pounds each, for both CCl_4 and TCE. The result of this calculation is directly dependent upon the R_f value that is chosen. As was shown in Table S-2, this parameter may be highly variable. The mass of CCl_4 and TCE in

the groundwater and sorbed to aquifer materials must be added to the residual DNAPL trapped in the aquifer material.

The estimated mass of residual DNAPL can only be very generally approximated because the spill volume is unknown. An estimate is needed because groundwater analytical data imply that residual saturations of DNAPLs, CCl_4 and TCE, are present in the aquifer. Assuming that residual DNAPL remains in 15% of the total porosity (estimated mean value of 0.40) of the saturated unconsolidated glacial materials on the site and that the DNAPL's downward extent after a single spill event is primarily within an area 10 feet by 10 feet with a depth of 125 feet, then potentially 750 ft^3 of DNAPL is present for each compound. Assuming an estimated 100 pounds per ft^3 of DNAPL indicates that residual DNAPL may contribute 75,000 pounds (34,020,000 grams) of contaminant mass on the site for each of the two compounds. Since the actual volume of the initial spills and the volume of residual DNAPL is not known, this approximation may be in error by several orders of magnitude. The total mass of chemical in each of the plumes would be approximately 38,577,000 (4,557,000 + 34,020,000) grams or approximately 85,000 pounds.

A mass of 85,000 pounds is equivalent to approximately 6,800 gallons of DNAPL. This volume is consistent with a volume that could be released from a single tank car spill and with the size of some of the past spills recorded at this site.

5.2.1.3 Chemical Loading to St. Joseph River

The current plume configuration has been delineated by analytical data obtained from groundwater samples collected from wells and LSA data. These data indicate a wide variation in chemical concentrations at the northwest edge of the site as monitored by downgradient well nests MW43, MW07, and MW08, and monitoring well MW09. Measured concentrations in these wells and nearby LSA data range from non-detectable to $820 \mu\text{g/L}$ for CCl_4 , and from non-detectable to $1,100 \mu\text{g/L}$ for TCE. Assuming that the flux of chemicals that passes through a cross-sectional area monitored by these downgradient wells is discharged completely to the St. Joseph River without loss or gain of chemical mass, then the data collected from these downgradient wells can be used to estimate the present loading of the chemicals from the plume to the river. The LSA sampling results can be used to extrapolate the monitoring well analytical data through the vertical extent of the aquifer.

The plume cross-sectional area where chemical flux is estimated is a plane oriented perpendicular to groundwater flow approximately parallel to MW43, MW07, MW08, and MW09. The flux is estimated through a plume width of approximately 3,000 feet and a depth of approximately 125 feet (i.e., a rectangle 375,000 ft² in size). Assuming the mean values used in previous estimates, an average hydraulic conductivity of 70 ft/day and horizontal groundwater gradient (i) of 0.002 ft/ft, the volume of groundwater passing through this rectangular area and discharging to the St. Joseph River is 52,500 ft³/day (or 1,500,000 liters/day) (discharge is Q, where $Q = KAi = (70 \text{ ft/day}) (375,000 \text{ ft}^2) (0.002 \text{ ft/ft})$). This estimate is sensitive to the boundary conditions used to formulate the calculation, and to the values chosen to execute the calculation. For example, the reference plane that is defined near well nests next to the St. Joseph River uses the analytical results from these wells at these locations as a basis for the concentrations of CCl₄ and TCE. If a reference plane located 1,500 feet upgradient from these wells is chosen, it would be located at MW38 and MW50. If the associated concentrations at this new location are assumed to discharge to the river, the estimated loading to the river would increase by an order of magnitude. The estimate is also directly dependent upon the value of the hydraulic conductivity. The chosen value for the hydraulic conductivity corresponds to the geometric mean of E & E Phase II slug test results. As shown in Table 3-3, the range of the Phase II slug test results is approximately plus or minus one order of magnitude. This would translate through the flux and loading calculations as a change of plus or minus one order of magnitude.

To estimate the chemical loading to the St. Joseph River, the variability of the chemical concentrations across the 375,000 ft² cross-sectional area of the plume over which the flux is being measured requires assigning concentrations to areas of the cross-section to obtain a weighted average. The estimate of the chemical loading is obtained by summing the mass of chemicals in the water that passes through this cross-sectional area as shown in Table 5-4. For this estimate it is assumed that the weighted average concentration for TCE and CCl₄ is the same. The result of this calculation may be a value that is biased low because concentrations of other contaminants present in the groundwater, such as CHL, are not included. The estimated loading from the site to the St. Joseph River is 20 pounds of TCE and 20 pounds of CCl₄ per year.

Water samples collected from the river during the Phase III RI did not reveal widespread contamination comparable to the groundwater contamination. This may be a reflection of the greater number of transformation processes operating in surface water systems compared to groundwater systems. In addition, the St. Joseph River has a discharge of 3,251 ft³/sec (or 7.95 x 10⁹ liters/day) (Stewart and Deiwert 1992). The relatively large discharge rate of the river suggests that dilution, a factor of 5,300, is a potentially dominant process.

5.2.1.4 Natural Attenuation

The estimated mass of dissolved CCl₄ and TCE remaining in the aquifer and the loading rate to the St. Joseph River can be the basis of a conceptual model used to estimate the time required for the dissolved contaminant mass to be discharged to the St. Joseph River. Assuming that equilibrium conditions exist across the site, current loading rates would be expected to remain relatively constant until the CCl₄ and TCE have been removed. Assuming that the combined loading of CCl₄ and TCE to the St. Joseph River is 40 pounds per year and is the only removal mechanism for the dissolved contaminant mass, the duration for removal may be estimated. The dissolved contaminant mass can be calculated using the following equation:

$$M_D = C n V,$$

where C is the concentration of the chemical in the groundwater, n is the total porosity, and V is the bulk aquifer volume (EPA 1989a). The calculation may be performed using the same methodology shown in Table 5-3. This yields a combined dissolved CCl₄ and TCE mass of 8,000 pounds. At a rate of 40 pounds per year it will take approximately 200 years to remove the dissolved CCl₄ and TCE from the aquifer. This conceptual model ignores many important factors that pertain to the entire existing contaminant mass. These factors would increase the time elapsed for natural attenuation to occur. They include the existence of pooled DNAPL, the existence of residual DNAPL, the amount of CCl₄ and TCE adsorbed to the aquifer material, and the amount of CCl₄ and TCE in the vadose zone (either sorbed or present as residual DNAPL).

An additional conceptual model could be used to estimate the time required for the entire contaminant mass to be discharged to the St. Joseph River. This model would involve flow of uncontaminated groundwater from locations that are upgradient of the site to the river. The contaminant-free groundwater would act to desorb contamination from aquifer material by continuously introducing fresh groundwater to the site. A calculation designed to estimate the time elapsed to remove the contamination would be extremely sensitive to the retardation factor and groundwater velocity values chosen. The ranges in these parameters have been presented in Table 5-2 and Table 3-3, respectively. Assuming a groundwater velocity of 200 feet per year and a retardation factor of 2.5, the time required for the center of mass of contamination to travel across the study area is 145 years. This estimate for natural attenuation is based on the same assumptions as the first conceptual model and is also biased low for the same reasons.

5.2.2 Site-Specific Transformation Processes

Daughter products of CCl_4 and TCE consistent with the anoxic reductive dehalogenation transformation processes shown in Figure 5-1 are detected in many of the groundwater samples. For example, 170 $\mu\text{g/L}$ of 1,2-dichloroethene (total) was detected in MW41D-02 and 15,000 $\mu\text{g/L}$ of TCE was present in the same sample. A concentration of 180 $\mu\text{g/L}$ of CHL was detected in MW38D-02, and 3,200 $\mu\text{g/L}$ of CCl_4 was also present. CHL was detected more frequently than 1,2-dichloroethene (total). This may indicate that high concentrations of CCl_4 are not necessary to initiate a small percentage of transformation to CHL. Methylene chloride, detected in MW46S-01 at 30 $\mu\text{g/L}$, is a daughter product of CHL. The concentrations of the daughter products are generally significantly lower than the concentrations of CCl_4 and TCE at the same locations. The daughter products and parent materials are all aliphatic chlorinated hydrocarbons, which will be transported similarly in the groundwater. This indicates that transformation is occurring at the site.

Table 5-1

PHYSICAL AND CHEMICAL PROPERTIES OF CHEMICALS OF POTENTIAL CONCERN

Chemical Name	CAS Number	Mole Weight (g/mole)	Physical State at 20° C	Water Solubility (mg/L)	Source	Liquid Density (g/mL)	Source	Vapor Pressure (mm Hg)	Source	Henry's Law Constant (atm-cm ³ /mole)	Source	K _{oc} (mL/g)	Source
Carbon tetrachloride	56-23-5	153.8	Liquid	8.00 E+02	H	1.59	A	115	+	2.41 E-02	I	110	+
Chloroform	67-66-3	119.4	Liquid	8.20 E+03	J	1.49	A	151	I	2.87 E-03	I	31	+
1,2-Dichloroethene (Total)	540-59-0	97.0	Liquid	3.50 E+03	B	1.28	C	208	D	7.58 E-03	E	49	D
Methylene Chloride	75-09-2	84.9	Liquid	2.00 E+04	H	1.33	C	349	H	2.03 E-03	H	8.80	II
Tetrachloroethene	127-18-4	165.8	Liquid	1.50 E+03	B	1.63	C	18.5	B	2.59 E-02	B	118	D
1,1,1-Trichloroethane	71-55-6	133.4	Liquid	1.50 E+03	B	1.35	C	123	B	1.44 E-02	B	152	D
Trichloroethene	79-01-6	131.4	Liquid	1.10 E+03	B	1.47	C	72.9	B	9.1 E-03	E	126	D

Key:

- A Verschuergen, K., 1983, *Handbook of Environmental Data on Organic Chemicals*, Van Nostrand Reinhold, New York.
 B Howard, P., 1989, *Handbook of Environmental Fate and Exposure Data for Organic Chemicals*, Volume 1, Lewis Publishers, Inc.
 C Wenz, R., ed., 1983, *Handbook of Chemistry and Physics*, 63rd edition, CRC Press, Cincinnati, Ohio.
 D Mabey, W., et al., 1982, *Aquatic Fate Process Data for Organic Priority Pollutants*.
 E Mackay, D., and W. Shiu, 1981, *J. Phys. Chem. Ref. Data*, 10:1175-1179.
 F Arthur D. Little, Inc., 1989, *The Installation Program Toxicology Guide*, Volumes 1 to 4.
 G Kinno, M., 1978, *Biodegradation and Bioaccumulation Tests on Chemical Substances*, OECD Tokyo Meeting, TSU-No. 3.
 H Knox, R.C., 1993, *Subsurface Fate and Transport Processes*, Lewis Publishers, Inc., Ann Arbor, Michigan.
 I Merz, J.W., 1990, *Basics of Pump-and-Treat Groundwater Remediation Technology*, EPA-600/8-90/003.
 J Schwille, P., 1988, *Dense Chlorinated Solvents in Porous and Fractured Media, Model Experiments*, Lewis Publishers, Ann Arbor, Michigan.

Table S-3

RETARDATION FACTORS

Contaminant	K_{oc} (ml/g)	f_{oc} Low Measurement	f_{oc} High Measurement	K_d ml/g using low f_{oc}	K_d ml/g using high f_{oc}	ρ_b Bulk Density g/cm ³	α Total Porosity (low estimate)	α Total Porosity (middle estimate)	α Total Porosity (high estimate)	R_f High Estimate of Retardation Factor using $\alpha = 0.30$ and $f_{oc} = 0.0230$	R_f Low Estimate of Retardation Factor using $\alpha = 0.50$ and $f_{oc} = 0.0230$	R_f Estimated Retardation for Major Groundwater Flow Zones Using $\alpha = 0.40$ and $f_{oc} = 0.0230$
Carbon tetrachloride	110	0.0000	0.0230	0.33	2.6	2.67	0.30	0.40	0.50	17.2	1.9	2.3
Chloroform	31	0.0000	0.0230	0.093	0.74	2.67	0.30	0.40	0.50	5.6	1.2	1.4
1,2-dichloroethane	49	0.0000	0.0230	0.15	1.2	2.67	0.30	0.40	0.50	8.5	1.4	1.6
Methylene chloride	8.8	0.0000	0.0230	0.026	0.21	2.67	0.30	0.40	0.50	2.3	1.1	1.1
Tetrachloroethane	118	0.0000	0.0230	0.35	2.8	2.67	0.30	0.40	0.50	18.4	1.9	2.4
1,1,1-trichloroethane	152	0.0000	0.0230	0.46	3.6	2.67	0.30	0.40	0.50	23.4	2.2	2.8
Trichloroethane	126	0.0000	0.0230	0.38	3.0	2.67	0.30	0.40	0.50	19.7	2.0	2.5

Notes:

 K_{oc} from Table S-1.High and low values for f_{oc} were obtained from Phase II Total Organic Carbon Analytical Results. $K_d = (K_{oc})(f_{oc})$ Bulk Density (ρ_b) assumed as equivalent to average specific gravity given in Table S-1.

Effective Porosity values obtained from Table S-3.

Retardation factor calculated using $R_f = 1 + \frac{(\rho_b)(K_d)}{\alpha}$ Freeze and Cherry (1979)where $\rho_b = \rho_s (1 - \alpha)$ = bulk density

Table 5-3
SIMPLIFIED ESTIMATE OF CHEMICAL MASS REMAINING IN AQUIFER

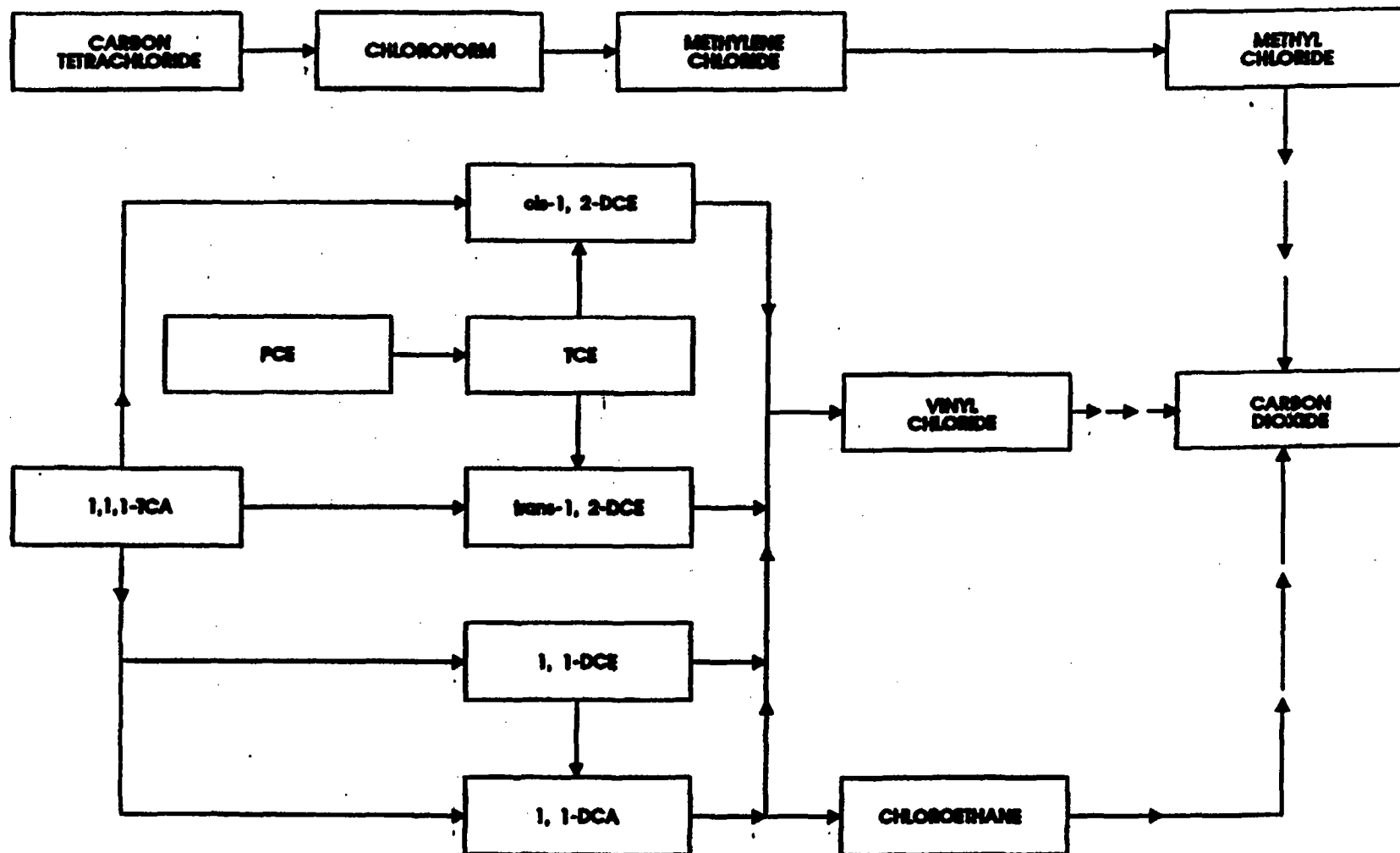
Retardation Factor (R_f)	Average Concentration for Volume Fraction (C) (in grams/liter)	Total Porosity (n) (given as fraction)	Volume (V)		Total Mass (M_T) sorbed and dissolved (in grams)
			Volume Fraction	Total Volume (in liters)	
2.5	0.1	0.40	0.0001	1.274×10^{11}	1,274,000
2.5	0.01	0.40	0.001	1.274×10^{11}	1,274,000
2.5	0.001	0.40	0.01	1.274×10^{11}	1,274,000
2.5	0.0001	0.40	0.0389	1.274×10^{11}	500,000
2.5	0.00001	0.40	0.15	1.274×10^{11}	191,000
2.5	0.000001	0.40	0.30	1.274×10^{11}	38,000
2.5	0.0000001	0.40	0.50	1.274×10^{11}	6,000
Totals			1.0		4,557,000

Note:

Where $M_T = R_f C n V$ = Total mass sorbed to aquifer materials and dissolved in groundwater (EPA 1989). To this mass, add estimated volume of residual DNAPL (or 34,020,000 grams) for a total estimated chemical mass of 38,577,000 grams or approximately 85,000 pounds each for CCl_4 and TCE remaining in the aquifer.

Table 5-4 GROUNDWATER CONTAMINANT MASS LOADING TO THE ST. JOSEPH RIVER			
Total Discharge of Groundwater Through Cross-Sectional Area (in liters/day)	Total Discharge with Estimated Chemical Concentration (given as fraction)	Estimated Concentration of Chemical (in grams/liter)	Grams of Chemical (grams/day)
1,500,000	0.01	0.001	15
1,500,000	0.04	0.0001	6
1,500,000	0.15	0.00001	2
1,500,000	0.30	0.000001	0.5
1,500,000	0.50	0.0000001	0.075

Note: Total loading to St. Joseph River is estimated at 25 grams per day or 20 pounds per year each for
 CCl_4 and TCE.



SOURCE: Adapted from Smith and Design, 1984.

FIGURE 5-1
TRANSFORMATION PATHWAYS FOR VARIOUS CHLORINATED ORGANICS IN
SOIL-GROUNDWATER SYSTEMS

6. RISK ASSESSMENT

A complete description of the Conrail railyard facility, including the site setting and the site background, can be found in Section 1 of this RI report. Section 4 describes the nature and extent of contamination detected at the Conrail site.

The objectives of this risk assessment for the Conrail Site are:

- to identify site-related contaminants of concern;
- to identify potential pathways of exposure for human and ecological receptors;
- to quantitatively estimate the exposures that could occur; and
- to estimate the potential risks to human health and the environment associated with these exposures.

At the direction of EPA, the human health evaluation (HHE) portion of this risk assessment will focus on the volatile organic contamination at the Conrail Site. Volatile organic compounds (VOCs) are mobile compounds because of their tendency to volatilize to the atmosphere or dissolve in water; they generally do not sorb to soil particles readily. Because of this mobility, there is a greater possibility for a larger population (i.e., both nearby residents and facility workers) to be exposed to these contaminants. Nonmobile compounds (i.e., semivolatiles, pesticides, PCBs, and metals) will not be addressed in this HHE; therefore, no conclusion will be made concerning the risks associated with these compounds in this HHE.

6.1 CONCEPTUAL SITE MODEL

A conceptual model for the site is presented in Figure 6-1 for current and potential future site conditions.

6.1.1 Current Site Conditions

Currently, most of the railyard is covered with railroad tracks and track ballast. A few structures are scattered throughout the facility. No Trespassing signs posted around the site are the only means of restricting visitor site access. Since there is very little exposed soil, and given the volatile nature of the contaminants detected on site, exposure of on-site receptors to surface soil contamination by direct contact is unlikely, and will not be evaluated in this HHE. On-site receptors, i.e., workers and visitors, however, could potentially be exposed to contaminants through the following pathways:

- Direct contact (dermal contact and incidental ingestion through hand-to-mouth contact) with contaminated subsurface soils; and
- Inhalation of vapors emanating from the ground surface as a result of volatilization of subsurface contaminants in site soils and groundwater.

Chemicals that have reached the groundwater can migrate with the groundwater, potentially affecting downgradient wells. There are two distinct plumes of contaminated groundwater migrating from the Conrail facility. The first plume (Plume 1) originates in the west central portion of the facility and is migrating north and west of the facility, whereas the other plume (Plume 2) emanates from the eastern portion of the facility and is migrating due north. Residential areas affected by contaminated groundwater in Plume 1 include: the County Road 1 residential area, the Charles Avenue residential area, and the Vistula Avenue residential area (see Figure 1-1 for the locations of these areas). The LaRue Street residential area (Figure 1-1) is the primary area affected by Plume 2. Residents with private wells in any of these areas could be exposed to groundwater contaminants through domestic use of groundwater. Potential exposure pathways for these residents include:

- Ingestion of drinking water.

- Dermal contact while showering, and
- Inhalation of vapors while showering.

Populations in the residential areas could also be exposed to contaminants through exposure to vapors in their homes. It is possible that VOCs detected in groundwater could volatilize into the surrounding porous soils, eventually migrating into subsurface structures (basements) and subsequently into the whole house.

Groundwater contaminants may eventually reach the surrounding surface water bodies. The groundwater beneath the railyard migrates north to the St. Joseph River and northwest to Bangor Bay. These water bodies are used for recreational purposes. Swimmers could be exposed to contaminants through dermal contact and by ingestion of surface water. Fishing also occurs in the river and the bay. In addition to direct contact with surface water, fishermen could be exposed to site-derived chemicals by consuming their catch. Nearby residents are likely to be the most frequent recreational users of surface water.

6.1.2 Future Site Conditions

The facility has been used as a railyard since the 1950s, and is expected to be used in this same capacity in the future. Given the economic importance of the railyard, it is extremely unlikely that the yard would be converted to residential or any other more sensitive use in the foreseeable future. It is also unlikely that residences would be constructed closer to the facility than they are presently located, which in some cases is across the street from the railyard.

6.1.3 Organization of the Risk Assessment

The HHE portion of this risk assessment has been prepared and organized in general accordance with EPA's *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual* (EPA 1989b), and other related or supplemental guidance. The ecological risk section is prepared and organized in accordance with EPA's *Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation Manual* (EPA 1989c).

The risk assessment is organized as follows. Sections 6.2 through Section 6.5 delineate the HHE. Sections 6.6 through 6.11 present the ecological risk assessment.

Section 6.1 discusses the site background and the human health conceptual site model. Section 6.2 reviews the site characterization data available, including sampling and analytical methods and data limitations, and identifies the chemicals of potential concern (COPCs) at the Conrail Site.

Section 6.3 assesses the potential exposure of human receptors to the COPCs. The potential exposure pathways are reviewed and exposure estimates are derived, taking into consideration the site setting and various site characteristics.

Section 6.4 provides toxicity assessments for the COPCs at the site. Toxicity assessment methodologies are reviewed and a brief discussion of the toxicological properties of each chemical is provided, along with tables summarizing the quantitative indices of toxicity for the COPCs.

Section 6.5 integrates the exposure and toxicity assessments from Sections 6.3 and 6.4 into an overall HHE. The primary potential risks associated with the site are identified, along with the pathways and chemicals giving rise to those risks.

Sections 6.6 through 6-11 detail the ecological risk resulting from site-related contaminants. These sections discuss the selection of contaminants of concern for ecological risks, the toxicological properties of these chemicals, and the potential ecological receptors, and finally present an evaluation of the risk to ecological receptors by integrating the information presented in earlier sections.

6.2 IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN

6.2.1 Data Collection

The objective of all three phases of the RI was to characterize the nature and extent of contamination associated with the Conrail Site, as well as the site topography, geology, and hydrogeology, in order to identify and evaluate potential migration and exposure pathways.

The investigative activities carried out to achieve this objective are described in Section 2 of this RI report. Environmental samples collected during the three phases of the Conrail RI/FS include: subsurface soils from the facility in areas of known or suspected contaminant sources; groundwater samples from monitoring wells installed during the three

phases of the RI/FS: and surface water and sediment samples from Baugo Bay, the St. Joseph River, and the Conrail ponds (located south of the classification yard). A discussion of what parameters the samples were analyzed for can be found in Section 2 of this report. The samples were analyzed using EPA CLP protocols as specified in the Phase I Conrail Site Quality Assurance Project Plan (QAPP) (E & E 1989b) and the Phase II and III Addenda to the QAPP (E & E 1991b, 1992c, respectively).

Air modeling of contaminant migration was performed to estimate exposure-point concentrations in ambient air and indoor air (private homes) based on measured groundwater and subsurface soil concentrations. Physical parameters measured during the RI, such as soil permeability, were used in the modeling calculations. Models were used to estimate the flux of contaminant vapors from the ground to ambient air, and a dispersion model was used to calculate downwind air concentrations. Grain-size distribution data and estimates of other soil properties derived from descriptions in boring logs were used to calculate contaminant emissions. Meteorological parameters used in the air dispersion model were obtained from the National Oceanic and Atmospheric Administration (NOAA) Stability Array (STAR) data for South Bend, Indiana.

Sampling of environmental media was performed using standard methodologies and quality assurance/quality control (QA/QC) procedures. The specific methods used are described in the RI/FS Work Plans (E & E 1991a, 1992b) and in Section 2 of this RI report.

6.2.2 Data Evaluation

6.2.2.1 Data Validation

Analyses of the RI samples were performed using methods and QA/QC procedures specified in the EPA-approved site QAPPs (E & E 1989b, 1991b, and 1992c). Analyses were performed by EPA CLP laboratories. Data validation was performed in accordance with EPA's functional guidelines for evaluating organic and inorganic analyses (EPA 1988a, 1988b, respectively) by EPA. The conclusion of the data quality review was that analyses were performed in compliance with the specific methods (except as noted for specific samples, see Section Appendix E of this RI report), and that the analytical results were acceptable for use as qualified.

For the purposes of the human health evaluation portion of the risk assessment, data from all three phases of the RI/FS were evaluated to determine potential risks to receptor populations. Specifically, data bases of groundwater and subsurface soil sample analytical results from all three phases in the investigation were compiled to evaluate potential risks.

6.2.2.2 Quantitation Limits

The data set used in this risk assessment combines the results of the three phases of site investigations conducted since 1989. Sample detection limits varied among samples and investigations due to differences in sample specific conditions and, in some cases, changes in the analytical methods used, as mandated by changes in the CLP laboratory Statement of Work (SOW), under which the contract laboratories operate. In accordance with EPA RAGS recommendations (EPA 1989b, c), the adequacy of the detection limits for soils/sediments and groundwater/surface water was evaluated by determining the risk associated with the lowest detection limit for each matrix.

Table 6-1 presents the typical sample detection limit or range of typical sample detection limits achieved for each chemical detected at the Conrail Site along with the corresponding cancer risk value and for noncarcinogenic compounds, the hazard index. Detection limits corresponding to a cancer risk greater than 10^{-6} or having an associated hazard index greater than 1.0, have been flagged in Table 6-1. For soil samples, the risks at detection limit concentrations were lower than the EPA risk levels. This indicates that the detection limits for soil are adequate for risk assessment purposes.

Groundwater sample detection limits for a number of chemicals, mainly carcinogens, exceeded the corresponding EPA risk levels. This indicates that these detection limits may not be entirely adequate for evaluating groundwater used directly as a residential water supply source. Significant risks could exist, but be overlooked if these chemicals were present in drinking water at concentrations below but approaching their detection limits.

6.2.2.3 Data Qualifiers

Several types of data qualifiers are associated with a number of the analytical values reported and validated by the data evaluation process. The most commonly encountered

qualifiers, their meaning, and their effect on the use of the data in the risk assessment are summarized in Table 6-2.

In accordance with the RAGS manual (EPA 1989b), if an analyte was found in a blank (B flag for organics), values for the corresponding samples were included in the risk assessment only if the sample value was more than 10 times the blank value for common laboratory contaminants or more than five times the blank value for other compounds.

Estimated values (E and J flags) and imprecise values (N, W, and * flags) were used because they are the best available estimates of the true concentrations present. The use of these values marginally decreases the accuracy and confidence in quantitative estimates of exposures and risks obtained by using them, which will be noted in discussions of uncertainties. Nevertheless, their use provided the best estimates obtainable.

U-flagged values, indicating that the chemical was not detected at the specified quantization limit, were evaluated on a case-by-case basis. If there was reason to believe the contaminant might be present, such as detection in a nearby sample, one-half of the sample quantization limit (SQL) was substituted for the U value, unless the value exceeded the maximum concentration of the chemical actually detected in the sample set. In that case, the U-flagged value was excluded from the quantitative risk assessment.

6.2.2.4 Blanks

Acetone, toluene, carbon disulfide, 2-butanone, and methylene chloride were found in some soil and groundwater samples. These chemicals are common laboratory contaminants and were also present in many of the blanks. In the validated data, sample concentrations less than 10 times the associated blank concentration were attributed to laboratory contamination and were discounted. Methylene chloride, however, is a breakdown product of chloroform, which is a known contaminant on the site. Therefore, special sample-specific consideration was given to methylene chloride. When methylene chloride was detected in samples that may have been affected by site-related chloroform contamination, professional judgement was used to determine whether the methylene chloride detected resulted from laboratory contamination or site-related contamination.

6.2.2.5 Background Concentrations

Volatile organic compounds usually are not naturally occurring chemicals, and are not generally found in samples from uncontaminated areas. Therefore, any VOCs detected in the background samples are from man-made sources. The only VOCs detected in the background groundwater and soil boring samples were laboratory artifacts, and were not addressed further in this HHE.

6.2.2.6 Previous and Current Phase Data Reporting

Monitoring wells were installed and sampled during all three phases of the Conrail Site field investigation. As a result, most Phase I wells were sampled three times, Phase II wells were sampled twice, and Phase III wells were sampled once. The sampling techniques used to collect the groundwater samples and the analytical methods used to analyze the samples were comparable throughout the three phases; therefore, monitoring well data from all three phases of the investigation were used in this HHE. To eliminate the possibility of weighing the data from the earlier-phase wells heavier than the later-phase wells—since the earlier-phase wells were sampled more times than the later-phase wells—data from each individual well was averaged to derive only one value for each chemical for each well. If a specific compound was undetected (flagged with a U) during one sampling round, and there was evidence that the compound was present in the well (i.e., it had been detected in earlier or later rounds of sampling), one-half the SQL was used in the averaging calculation.

Soil boring data from all three phases were also used in this HHE. However, no averaging was necessary since data were not collected from the same boring more than once. For purposes of this HHE, only contaminants detected in soil samples collected between 0.5 feet to 12 feet BGS were evaluated. Contaminants at deeper depths are unlikely to directly pose risks to human health. Also, contamination in deeper soils is indirectly evaluated via the groundwater pathways.

6.2.2.7 Selection of Chemicals of Potential Concern

For the HHE, COPCs were selected based primarily on their site-wide frequency of detection and the levels at which they were detected. Tables 6-3 through 6-5 present the frequency of detection and the range of concentrations detected for each volatile chemical

detected in subsurface soils (0.5 to 12 feet BGS), groundwater, and surface water and sediments, respectively. As explained above, only CLP data were used in the COPC selection process. Chemicals detected at low frequencies (i.e., detected in less than 5% of the samples) or at low concentrations (based on a qualitative comparison to toxicity values) were excluded from further consideration in the risk assessment because they are not representative of site conditions and are unlikely to contribute significantly to overall risks at the site. However, no chemicals were eliminated that are considered carcinogens, primary contaminants, or degradation products of these contaminants (see Figure 5-1), or that constituted a hot spot of contamination.

6.2.3 Summary of Analytical Results and Chemicals of Potential Concern

The VOCs identified in various environmental media sampled during three phases of the Conrail Site field investigation are summarized in Tables 6-3 through 6-5 as follows:

- **Soils (0.5 to 12 ft BGS) - Table 6-3;**
- **Groundwater - Table 6-4; and**
- **Surface water and Sediments - Table 6-5.**

The tables present the frequency of detection and the range of concentrations detected for each chemical. The data are reported in these tables without laboratory or data validation qualifiers since they are the best estimates of the true concentrations of the chemicals detected in the samples. For a list of the data with qualifiers, see Appendices B and E of this report.

For groundwater, the detection frequencies shown in Table 6-4 are based on the total number of wells sampled, which is 64, although generally three samples were taken from Phase I wells, two samples from Phase II wells, and one sample from Phase III wells. If a contaminant was observed more than once in a well, it was recorded as being detected in that well once.

The chemicals detected in site soils and groundwater were screened as described in Section 6.2.2.7 in order to select COPCs. Next to each chemical name in these tables is a footnote marker that designates it as a COPC or explains why the chemical was eliminated. COPCs in site soil and groundwater are summarized in Table 6-6. Of the three groundwater

COPCs that are not soil COPCs (i.e., chloromethane, 1,1-dichloroethane, and 1,1-dichloroethene), none was detected in any subsurface soil sample (0.5 to 12 feet BGS).

Generally, the chemicals detected in surface waters and sediments are subsets of the chemicals detected in site soils and groundwater. The risks from exposure to these chemicals are minimal (e.g., less than 5% of the total risk) due to the low concentrations detected and the toxicity values of the compounds detected. Therefore, they were not evaluated in the HHE.

6.3 EXPOSURE ASSESSMENT

6.3.1 Physical Setting

The physical setting of the site is described in detail in Section 1 of this RI/FS report.

Potentially Exposed Populations

Most of the classification yard is covered with railroad ballast (approximately 2 inch diameter crushed stone) and tracks, and only small patches of soils are exposed. This, in addition to the fact that the COPCs are volatile and have little residence time in the surface soils, results in facility workers and visitors having little opportunity to come into direct contact with COPCs in the surface soils. Railyard workers and site visitors, however, may be exposed to vapor emissions (volatilization) from subsurface sources, namely contaminated subsurface soils and groundwater.

The St. Joseph River is located approximately 1/2 mile north of the Conrail facility. Throughout the area south of the river and north of the railyard, groundwater flows towards the river. The river may be contaminated due to this discharge of contaminated groundwater. The river is used extensively for recreation (boating, swimming, and fishing) and commercial boat travel, but it is not a drinking water source in the vicinity of the site.

The nearest residences are located approximately 50 feet north of the facility. Residential areas continue from this location to the St. Joseph River. Groundwater is used in this area as a domestic water supply, with water being drawn from the contaminated aquifer. Additionally, homes in this area may have basements that could be infiltrated by vapors from contaminated groundwater. These vapors could eventually disperse throughout the house.

leaks or spills of these materials. These past spill events and other areas of potential concern in the Conrail railyard are detailed in the RI/FS work plans (E & E 1989a, 1991a, and 1992b), and were the subject of this RI/FS investigation and associated risk assessment.

The number of spills and leaks, and the total volume of contaminants released are unknown. Information on the purity of the chemicals spilled or leaked is also not known. A discussion of reported spills and areas of concern can be found in Section 1 of this RI report.

For purposes of this risk assessment, and since all the sources of contamination in the railyard have not been identified, the facility will be treated as one large source area. This is a reasonable assumption since railyard workers may work in several areas throughout the work day and do not generally concentrate their activities in any one area.

6.3.2.2 Contaminant Fate and Transport

A detailed discussion of the fate and transport of the major site contaminants in the environment is presented in Section 5 of this RI report. Additional information concerning the effects of site-specific factors is summarized briefly in this section.

The VOCs, which have moderate to high vapor pressures, low to moderate water solubility, and little tendency to absorb to soil and sediment, are highly mobile in the environment. At the surface, VOCs rapidly volatilize to the atmosphere. In subsurface soils they can diffuse via soil gas to the surface or migrate downward with infiltration of precipitation, eventually reaching groundwater.

Persistence of the COPCs found at the site is influenced by soil moisture content, soil organic carbon content, the presence of microbiological populations capable of degrading the contaminants, and the availability of necessary nutrients and environmental conditions for microbial activity.

Many of the organic contaminants reportedly undergo biotransformation or biodegradation in the soil or groundwater under various environmental conditions. Benzene and alkylbenzenes (toluene, xylenes, ethylbenzene, etc.) undergo oxidative degradation under aerobic conditions (Verschuere 1983). Wilson, Smith, and Rees (1986) have shown that these compounds can also undergo reductive decomposition under anaerobic, methanogenic conditions, although lag periods of up to five months or more may precede significant biodegradative activity under anaerobic conditions.

Chlorinated methanes (chloroform, methylene chloride), chlorinated ethenes (tetrachlorethane, trichloroethene, dichloroethenes, and vinyl chloride), and 1,1,1-trichloroethane undergo sequential reductive dehalogenation under anoxic conditions, according to the scheme shown in Figure 5-1 (Smith and Dragun 1984). Chlorinated alkenes, however, do not degrade in oxygenated water (Wilson and McNabb 1983). Vinyl chloride may accumulate as the final product of this process if other carbon sources are present (Vogel and McCarty 1985), but, in the absence of such sources, vinyl chloride may be further degraded and ultimately mineralized (Parsons, Wood, and DeMarco 1984; Kleopfer *et al.* 1985; Wilson, Smith, and Rees 1986).

The biodegradative processes described are potential removal mechanisms for the organic contaminants identified in environmental media at the site; however, significant contaminant removal through these mechanisms is unlikely to occur unless the environmental conditions are favorable. The first requirement for significant removal is that a microbial population capable of degrading the specific contaminants is present at a sufficiently high density or that conditions exist that will support growth of such a population. The native microbial population density normally decreases sharply with increasing depth, and the population present at 30 feet to 50 feet BGS often is not large enough to make biodegradation a significant removal mechanism. The presence of a contaminant that can serve as a carbon source can sometimes induce a growth phase in a microbial population able to utilize the source, but only if the contaminant is present at a sufficient concentration for an adequate period of time. Even if an adequate microbial population and a sufficient contaminant concentration exist, other environmental factors, such as oxidizing or reducing conditions and an adequate supply of other nutrients, must be available for significant biodegradation to occur. Frequently, one or more of the necessary conditions is lacking, particularly deeper in the subsurface and, as a result, significant biodegradation may not occur spontaneously.

Information obtained during the RI suggests that biodegradation may be a significant fate process for some organic contaminants at the Conrail Site. A number of compounds are present in the subsurface soil and groundwater that are known degradation products of other compounds found at the site. For example, 1,2-DCE and vinyl chloride are degradation products of TCE, which itself may be a degradation product of tetrachlorethane; 1,1-DCA and

chloroethane are degradation products of 1,1,1-trichloroethane; and chloroform is a degradation product of carbon tetrachloride.

6.3.2.3 Potentially Complete Exposure Pathways

Potentially complete exposure pathways under current and potential future site conditions are presented schematically in Figure 6-1.

Under current conditions, facility receptors (workers and visitors) could potentially be exposed to contaminant vapors emanating from the ground surface as the result of volatilization of contaminants in subsurface soils and groundwater. It is unlikely that these receptors would come in direct contact with contaminated surface soils primarily due to the lack of exposed soils (most of the site is covered with tracks and track ballast) and due to the volatile nature of the contaminants. Under current conditions, facility workers could potentially be exposed to contaminants by inhalation of vapors emanating from the ground surface as a result of subsurface contamination of soils and groundwater.

Also, due to the physical conditions at the site (i.e., the lack of exposed surface soils), it is unlikely that contaminants can be transported off site by wind dispersion of particulates. However, contaminants can be transported off site in groundwater, potentially affecting downgradient wells. Currently, the groundwater of primary interest is the unconfined aquifer in the glacial outwash sands north and west of the Conrail facility. Residential, business, and light-industrial areas surround the site and have individual private wells and private septic tanks. An Interim Remedial Action is ongoing to provide residences downgradient of the facility (north and west) with service connections to the public water supply. Some nearby residences have had bottled drinking water or carbon filters supplied until the Interim Remedial Action is complete. Residents could potentially be exposed to groundwater contamination through domestic water usage by the following exposure routes:

- Ingestion of drinking water;
- Dermal contact while showering; and
- Inhalation of vapors while showering.

Additional residential exposures to site contaminants could occur from exposure to contaminated indoor air in their homes. Vapors emanating from contaminated groundwater can volatilize into subsurface soils, infiltrate residential basements and subsequently disperse throughout the house.

Groundwater from the site discharges to the St. Joseph River and Baugo Bay. For swimmers, potential exposures to site-related contaminants could occur by:

- Incidental ingestion of surface water, and
- Dermal contact with surface water.

Local recreational fishermen and their families could potentially be exposed to contaminants from the river and bay by:

- Incidental ingestion of surface water while fishing; and
- Consumption of fish from these water bodies.

However, swimmers and fishermen utilizing the river and/or the bay in areas that may be affected by COPCs are most likely local residents. These residents are also exposed to COPCs via the pathways discussed above. The risks from fishing and swimming in the river and bay are less than 5% of the total risk posed to these residents via all pathways. Therefore, the recreational pathways of swimming and fishing in the river and bay will not be evaluated further in this HHE.

Future receptors will likely remain the same as current potential receptors outlined above. It is unlikely, given the economic factors associated with a railyard of the size and importance of the Conrail facility, that future use of the Conrail facility would be anything other than as a railyard. Railyard workers, however, could come in direct contact with contaminants when digging or moving soils during future excavation activities (perhaps digging to install or repair underground pipes or cables). If this occurs, contaminated soils that are presently below the ground surface (i.e., 0.5 feet to 12 feet BGS), could become exposed, and should be evaluated as though they were surface soils. Therefore, a future

railyard excavation scenario will be evaluated. This scenario will include the determination of risks from exposures through the following pathways:

- Dermal contact with newly exposed contaminated subsurface soils;
- Incidental ingestion by hand-to-mouth contact with newly exposed contaminated subsurface soils; and
- Inhalation of vapors from newly exposed contaminated subsurface soils.

The potentially complete exposure pathways and receptors selected for quantitative evaluation are summarized in Table 6-7. This table also lists the data and the models that were used to derive exposure-point concentrations for these pathways.

6.3.3 Quantification of Exposure

This section describes how the quantitative exposure estimates were obtained for the pathways outlined in Table 6-7. Section 6.3.3.1 describes how the exposure-point contaminant concentrations used in the exposure assessment calculations were selected or derived. Section 6.3.3.2 describes the contaminant migration models that were used to estimate exposure-point concentrations, and Section 6.3.3.3 describes the exposure estimation calculations for each receptor and route of exposure.

6.3.3.1 Source Media Contaminant Concentrations

Table 6-7 summarizes the exposure points and exposure pathways evaluated for each receptor as well as the basis and derivation of exposure-point concentrations. The table also shows that soil and groundwater data from different areas were used to derive estimates of exposure-point concentrations.

The 95 percent upper confidence limit (95 UCL) on the arithmetic mean, or the maximum observed value (if the 95 UCL exceeded the maximum value or if fewer than 10 samples were available for the calculation), was used to estimate the subsurface soil concentrations used for both the reasonable maximum exposure (RME) case and for the central tendency (CT) case (EPA 1992b). Table 6-8 lists the maximum observed soil

concentrations and 95 UCL for each chemical detected. Prior to determining the 95 UCL, it was necessary to determine whether the data were normally distributed. Figure 6-2 shows the distribution of two of the soil COPCs. All subsurface soil samples (0.5 to 12 feet BGS) were used to calculate the 95 UCL regardless of location on the facility. As evidenced by the graphs in Figure 6-2, the soil COPC concentrations are not normally distributed, which may be due to, among other things, the presence of more than one source area on the site. Therefore, prior to calculation of the 95 UCL, all the soil data were log-transformed per the *Supplemental Guidance to RAGS: Calculating the Concentration Term* (EPA 1992b).

Groundwater concentrations were determined for both the RME and CT cases by determining the maximum concentration of each contaminant individually (after averaging individual well results to arrive at one value per COPC per well) in monitoring wells located in Plume 1 residential areas (Conroy Road 1 areas, Charles Avenue areas, and Vistula Avenue areas) and in monitoring wells located in the Plume 2 residential area (LaRue Street area). This provides a conservative (health-protective) estimate of the concentrations that residents and workers could be exposed to.

The residents of neighborhoods north of the facility could potentially be exposed to facility-related chemicals in their drinking water, which comes from the aquifer in the glacial outwash sands north and west of the facility. Private drinking water wells could be installed at any depth in either of the plumes emanating from the railyard. Therefore, the maximum concentration detected for each COPC in the plumes was used to estimate exposure from drinking water ingestion, dermal contact while showering, and vapor inhalation while showering. Drinking water exposures were estimated for residents in each plume based on maximum concentrations of COPC in monitoring wells in each plume. A shower volatilization model was applied to the maximum groundwater concentrations to calculate vapor concentrations in the shower stall.

Indoor air vapor exposure-point concentrations for each COPC were estimated using the maximum concentration of each COPC detected in the shallow monitoring wells in each groundwater plume. Only the concentrations from the shallow wells were used to estimate volatilization of the COPCs. COPCs detected in intermediate and deep wells would have minimal impact on soil gas volatilization rates. Vapor migration models were used to

estimate volatilization of COPCs from contaminated groundwater through the porous soil, into nearby basements and subsequently throughout the entire home.

Railyard worker exposures were evaluated for the whole facility. Data from subsurface soil borings (0.5 feet to 12 feet BGS) in conjunction with data from monitoring wells located on the facility were used along with subsurface vapor migration and air dispersion models to estimate the exposure-point concentrations of vapors in air on the facility. Again, whole facility exposures were calculated since work occurs throughout the facility, and exposures could happen anywhere contaminants are present below ground.

Dermal contact, incidental ingestion, and inhalation of vapors from newly exposed contaminated soils could occur during future excavations, which could occur anywhere on the facility. Concentrations from soil samples collected 0.5 feet to 12 feet BGS were used to estimate exposure-point concentrations for subsurface soil exposures (soil excavations).

6.3.3.2 Summary of Contaminant Migration Modeling Methods

This section briefly describes the modeling methods that were used to estimate exposure-point concentrations in ambient air and indoor air.

Each of the models described below was applied to estimate exposure concentrations for both the RME and CT exposure scenarios. The basic input to the models was the 95 UCL on the mean or the maximum observed concentrations of subsurface soil samples (whichever was lower), or the maximum observed groundwater concentrations. These models are described in detail in Appendix G.

Vapor Emissions from Soil

Volatilization of contaminants from unpaved, contaminated soil areas within the facility was estimated using the Farmer model (Farmer *et al.* 1978), which is recommended by EPA for Superfund applications (EPA 1988c) and by other modeling guidance sources (Gas Research Institute [GRI] 1988). This model essentially describes the molecular diffusion of vapors from the subsurface upward through the porous soil layer to the surface. The Millington-Quirk equation for the effective diffusion coefficient was used in this model to account for the effect of soil moisture.

A slight modification of the Farmer model was made to relate the concentrations in the subsurface soil air-filled pore spaces, which is the "driving-force" for diffusion in the model, to the measured contaminant concentrations in the subsurface areas. Rather than unconditionally assuming that the soil pores are saturated with vapor, as assumed in the original application of the Farmer model, the soil pore-space concentrations were estimated from the measured contaminant concentrations in subsurface soil or in groundwater, using equilibrium-partitioning relationships, i.e., Henry's Law or the soil-water partition coefficient, K_d . This partitional approach is used in other EPA guidance (EPA 1986a), and is also described in Thibodeaux (1979) and GRI (1988).

Concentrations of contaminants detected in soils 0.5 feet to 12 feet BGS and in groundwater from wells located on the facility were considered to represent a single source of contaminant vapors. Therefore, to avoid double-counting vapor emissions from saturated soils and groundwater, the total volatilization flux at the ground surface was calculated using both the groundwater and the subsurface soil concentration for each individual COPC. Whichever concentration yielded the highest flux (i.e., the groundwater or subsurface soil emission) was the concentration used to calculate potential risks. Use of the maximum flux value provides the most conservative estimate of exposure-point concentrations, and resultantly, the most conservative risk. Airborne vapor concentrations in the breathing zone were estimated using the estimated vapor emission rates from the ground surface and a near-field box model (GRI 1988).

Indoor Air Concentrations

Contaminant concentrations in indoor air resulting from infiltration of vapors from contaminated groundwater were estimated using a three-step process. The contaminant concentrations in the soil gas immediately above the groundwater were assumed to be in equilibrium with the groundwater and were calculated using the Henry's law constants. Attenuation of the vapor concentrations between the groundwater and building basements, due to adsorption and capillary fringe effects, was estimated empirically by comparing the soil gas concentration at the groundwater surface predicted by the Henry's law relationship with measured soil gas concentrations at the same location but at a shallower depth. Finally, attenuation of contaminant concentrations between the soil gas adjacent to a building's

basement and indoor air was estimated using an attenuation factor based on the Johnson and Ettinger model for infiltration of chemical vapors into buildings through cracks and openings in the foundation described in EPA 1992. Details of this model are presented in Appendix G.

6.3.3.3 Exposure Estimation Methods

The exposure estimates described in this section combine the following:

- Estimates of exposure media (subsurface soil, groundwater, and air) *contaminant concentrations developed in the previous two sections;*
- Estimates of contact rate and the frequency and duration of exposure that receptor populations are likely to experience; and
- Estimates of various physiological parameters (e.g., breathing rate, body weight, and average life expectancy).

The equations used to estimate the exposure for each pathway and route of exposure are given in Tables 6-9 through 6-16 in this section. The parameter values used to evaluate the equations along with the rationale for their selection and a reference source are also given.

For each of the exposure scenarios, parameter values were selected to correspond to the CT exposure and the RME that an individual in the receptor group might experience. The exposure point concentration used for both the CT and for the RME case was 95 UCL on the mean or the maximum observed concentrations of subsurface soil samples (whichever was lower), or the maximum observed groundwater concentrations. In most cases, standard default exposure factors from EPA's *Supplemental Risk Assessment Guidance for Superfund* (EPA 1991a) were used. Exposure factors not specified in this document were taken from EPA's *Exposure Factors Handbook* (EPA 1989d) or were based on professional judgment.

For the first exposure route, all of the parameters will be described and discussed in the text below; for subsequent routes, only the key parameters for that route and parameters not previously mentioned will be described.

EXPOSURES POTENTIALLY OCCURRING UNDER EXISTING CONDITIONS

Scenario 1: Facility Worker Exposure

The current facility worker exposure was evaluated for the Conrail facility as one source, and is presented below.

Pathway 1A: Inhalation of Airborne Vapors (Table 6-9). The inhalation rate (IR) is the receptor's estimated breathing rate for a moderate level of activity (EPA 1991a). The exposure time (ET) is assumed to be a normal workday, i.e., 8 hours per day. The exposure frequency (EF) was assumed to be 250 days per year for both the CT case and for the RME. The exposure duration (ED) was assumed to be 5 years for the CT case (United States Department of Labor [USDL] 1987) and 25 years for the RME (EPA 1991a). The body weight (BW) is the average for an adult male, 70 kg (EPA 1991a). Averaging time (AT) is the period over which the estimated exposure is averaged. For noncarcinogens, the averaging time is equal to the exposure duration, whereas for carcinogens, it is equal to the standard life expectancy of 70 years because the carcinogenic potency slope factors (see section 6.4.3.4) are based on lifetime exposure.

Scenario 2: Residential Exposure to Groundwater Contaminants

Residential exposure to groundwater contaminants may occur in either of the two plumes emanating from the facility, and the exposures will differ depending on the plume. Therefore, separate exposure assessments will be made for the residents exposed to contaminants in Plume 1 (County Road 1 area, Vistula Avenue area, and Charles Avenue area) and for the residents exposed to contaminants in Plume 2 (LaRue Street area). The exposure factors used for each plume will be the same, however, and are presented below. The only difference between the assessment of the two plumes will be the specific contaminants and concentrations used in the assessment. The Plume 1 assessment will include the maximum concentrations of those chemicals that are found in Plume 1, whereas the Plume 2 assessment will be performed with the maximum concentrations of those contaminants found in Plume 2.

Pathway 2A: Ingestion of Chemicals in Drinking Water (Table 6-10). The IR values used were the upper 90th percentile values for the RME case (EPA 1991a) while the CT value is the average intake observed in five studies (EPA 1993b). The EF represents year-round exposure allowing for two weeks away from the area. The values used for the exposure duration (ED) correspond to the 50th percentile and 90th percentile amount of time spent living at one residence for the CT exposure case and the RME case, respectively. The body weight (BW) is the average for an adult male (EPA 1991a).

Pathway 2B: Dermal Contact with Chemicals while Showering (Table 6-11). The values used for skin surface area are average and maximum values of a range of total body areas. The permeability constant (PC) is chemical-specific. PC values were obtained from Table 5-7 or Equation 5.8 of EPA's *Dermal Exposure: Principles and Applications* (EPA 1992c). Table 6-17 contains a list of the dermal permeation constants used for chemicals evaluated in this pathway. The value used for ET is recommended in the *Exposure Factors Handbook* (EPA 1989d) for evaluating this exposure route.

Pathway 2C: Inhalation of Volatile Chemicals While Showering (Table 6-12). The contaminant concentration in air was modeled from the water concentration as described in Appendix G. The other variables for this pathway have been described above.

Pathway 2D: Inhalation of Indoor Airborne Vapors (Table 6-13). The indoor air contaminant concentration was modeled from the groundwater concentration and subsequent soil vapor concentration as described in Appendix G. The other variables for this pathway have been described above.

EXPOSURES POTENTIALLY OCCURRING UNDER ALTERNATE FUTURE SITE CONDITIONS

Scenario 3: Future Facility Worker Exposure

A future facility worker scenario was envisioned in which exposures to soils up to 12 feet BGS could occur as a result of excavations that might occur during various construction or maintenance activities. Risks from exposures to COPCs (VOCs) will be evaluated in these

Exposure Estimates

The exposure estimates from the equations described above are given as lifetime average daily intakes (LADIs) for carcinogenic effects and as chronic daily intakes (CDIs) or subchronic daily intakes (SDIs) for noncarcinogenic effects for each complete pathway and exposure case in the risk estimation tables in Appendix H. The exposure estimates are combined with toxicity estimates for each chemical discussed in Section 6.4 to obtain risk estimates.

6.3.4 Uncertainty in the Exposure Assessment

A number of factors will cause the exposure levels estimated in the exposure assessment to differ from the exposures that potential receptor populations may actually experience. This section will identify these factors, discuss the potential effects of the factors on the exposure estimates, and, where possible and appropriate, estimate the degree of confidence that should be placed in the various assumptions and parameter estimates that have gone into the exposure estimates.

6.3.4.1 Environmental Sampling and Analysis

Samples collected during the three phases of the Conrail site field investigation were intended to characterize the nature and extent of contamination at the site. Accordingly, most samples were collected from locations selected in a purposeful, directed manner to accomplish this goal. Samples collected in this manner provide considerable information about the site but are not statistically representative of the contamination that may be present on the site as a whole. In order to gather statistically representative data, the sampling locations need to be selected in a random fashion or through the use of a designated grid system. Few of the sampling locations in the risk assessment data base were selected in this way. Sampling locations selected in a directed manner tend to be concentrated in areas having higher levels of contamination; therefore, data from sampling locations selected in this way tend to overestimate the average concentrations present in a representative exposure area.

Soil and groundwater chemical analyses were focused primarily on VOCs. VOCs were evaluated because they are highly mobile, and therefore there is a greater possibility for human receptors to be exposed to these contaminants. This analytical focus may have

6.3.4.4 Steady-State Assumptions

Most of the exposure calculations used in this HHE assume that the concentrations of chemicals in the source media are at steady state and remain constant for the duration of the exposure periods, which range from five to 30 years. Only the future worker inhalation pathway assumes source depletion over time.

This steady-state assumption may not be appropriate for organics in the soil, which tend to migrate, eventually depleting the source, or may degrade over time. The concentrations of contaminants in groundwater could increase or decrease over the next 30 years, depending on many site-specific factors. For simplicity, the conservative (health-protective) steady-state assumption was used in all but the one pathway listed above.

6.3.4.5 Contaminant Migration Modeling

Uncertainties about the reliability of modeling predictions arise in two areas. The first is the validity and accuracy of the model itself, and the second is the selection of appropriate parameter values for use in the calculations. Several types of models were used in developing exposure estimates.

One model concerned the emission of vapors from surface soil. The modified Farmer model, described in *The Superfund Exposure Assessment Manual* (EPA 1988c), was used to estimate the volatilization flux of VOCs at the ground surface from subsurface contamination. The selection of conservative values for some parameters used in this model (soil porosity, soil-water partition coefficient, size of source area, etc.) will tend to overestimate the actual emissions, but the results should be moderately reliable.

Air dispersion modeling is used in conjunction with the model described above to estimate ambient air concentrations downwind from the source areas. The near-field box model (GRI 1988) was used to estimate air concentrations of COPCs. Standard regulatory default options were used as model inputs along with meteorological data from the airport in South Bend, Indiana. The results of this model should be moderately reliable.

The third type of model was used to estimate indoor air concentrations. An empirical model was employed to calculate potential concentrations of vapors infiltrating basement walls from the surrounding porous subsurface soils with subsequent dispersion throughout the home. The soil gas concentrations adjacent to basements were calculated from the

groundwater concentrations using a simple partitioning equation employing the Henry's Law Constant and an empirical attenuation factor. Standard regulatory default values, which tend to be conservative (health-protective), were used in this empirical model. This may result in overestimating the vapor concentrations, but the results should be moderately reliable.

Taken together, the results for all the models employed should be moderately to highly reliable, but may overestimate actual exposure-point air concentrations.

6.3.4.6 Exposure Assessment Uncertainty Summary

Overall, the exposure estimates obtained are probably moderately reliable. Several of the factors adding uncertainty to the estimates tend to result in overestimation of the exposure. These include:

- The directed nature of the sampling and analysis program;
- The use of the 95 UCL or the maximum observed value for the source concentrations;
- The use of a number of 90th percentile values in the exposure estimation calculations;
- The use of the steady-state assumption for source concentration estimates; and
- The use of transport models, which incorporate their own conservative assumptions, to estimate exposure-point concentrations in ambient and indoor air.

VOCs were the only group of contaminants evaluated in this HHE. By not evaluating other contaminants that may exist at the site, risks to humans may be underestimated because of the potential exposure pathways of contamination that were not evaluated.

The use of detection limits for chemicals in water analyses that were not entirely adequate for risk assessment purposes could lead to underestimation of exposures to these chemicals through the direct use of groundwater. The cumulative effect of all exposure uncertainties, however, is most likely to overestimate rather than underestimate the true potential exposures.

6.4 TOXICITY ASSESSMENT

6.4.1 Introduction

The purpose of the toxicity assessment is to compile toxicity and carcinogenicity data for the chemicals of potential concern at the Conrail Site and to provide an estimate of the relationship between the extent of exposure to a contaminant and the likelihood and/or severity of adverse effects. The toxicity assessment will be accomplished in two steps: hazard identification and dose-response assessment.

Hazard identification is a qualitative description of the potential toxic properties of the chemicals of potential concern at the site. Brief health effects summaries for the chemicals of potential concern are also presented.

The dose-response evaluation is a process that results in a quantitative estimate or index of toxicity for each contaminant at the site. For carcinogens, the index is the slope factor (SF). For noncarcinogens, it is the reference dose (RfD). Practices and procedures used to develop quantitative indices of toxicity and to incorporate toxicological information into the risk estimation process and the quantitative indices of toxicity are presented in Section 6.4.3. Uncertainties in the toxicity assessment process are discussed in Section 6.4.4.

6.4.2 Health Effects Summaries

The health effects summaries describe the potential toxic properties of the chemicals of potential concern at the Conrail Site. For carcinogens, the weight-of-evidence category is also included. In most cases, the information in the summaries is drawn from the Public Health Statement in the Agency for Toxic Substances and Disease Registry's (ATSDR's) toxicological profile for the chemical (ATSDR 1989-1993). Exposure concentrations that may be associated with adverse effects are included in some of the summaries.

Acetone

Acetone is a clear, colorless liquid with a cool, mint-like odor. It is a naturally occurring, volatile metabolite that has been identified in such plants as onions, grapes, apples, tomatoes, and the morning glory. Acetone is a component of human breath and can also be emitted from natural sources during volcanic eruptions and forest fires. It is used as a solvent and as a chemical intermediate.

Most acetone used as a solvent or emitted from natural sources will ultimately be released to the air. If released to soil, it can both volatilize and leach to groundwater. Acetone also biodegrades readily.

The most likely human exposure to acetone is occupational. It is possible that the general population could be exposed to acetone, however, through inhalation of automobile exhaust, solvent vapors, and tobacco and wood smoke; through dermal or inhalation contact with consumer products such as nail polish remover; or by ingestion of food or water that contains acetone.

In high concentrations or in pure form, acetone is a skin irritant and severe eye irritant. Following inhalation exposure, nasal irritation, eye irritation, respiratory tract irritation, nausea, vomiting, and muscle weakness can occur. Kidney damage, metabolic changes, and coma can result from ingestion of large amounts of acetone. Increased liver and kidney weights and kidney toxicity are the critical, or most sensitive toxic effects, of acetone in animals and serve as the basis of EPA's RfD.

Acetone is not classified by EPA as to its carcinogenic potential.

Benzene

Benzene, in the environment, comes from both natural processes and human activity. Today, most benzene is produced from petroleum sources. Benzene has a long history of industrial use, most notably as a solvent and as a starting material for the synthesis of other chemicals.

Benzene evaporates easily, and exposure of the general public to benzene occurs mainly by breathing contaminated air. The major sources of benzene in air are gasoline and automobile exhaust, tobacco smoke, and industrial emissions. It has been estimated that 50 percent of the exposure to benzene in the United States is due to tobacco smoke. Household products including glues, paints, furniture wax, and detergents can also be a source of exposure.

Benzene is readily absorbed by inhalation and ingestion, but is absorbed to a lesser extent through the skin. Most of what is known about the human health effects of benzene exposure is based on studies of workers who were usually exposed for long periods to high concentrations of benzene.

Benzene is toxic to blood-forming organs and to the immune system. Excessive exposure (inhalation of concentrations of 10 to 100 ppm) can result in anemia, a weakened immune system, and headaches. Occupational exposure to benzene may be associated with spontaneous abortions and miscarriages (supported by limited animal data), and certain developmental abnormalities such as low birth weight, delayed bone formation, and bone marrow toxicity. Benzene is classified as a Group A, human carcinogen, based on numerous studies documenting excess leukemia mortality among occupationally exposed workers.

Bromodichloromethane (BDCM)

Bromodichloromethane (BDCM) is a colorless, heavy, nonburnable liquid. Most BDCM in the environment is formed as a byproduct of the addition of chlorine to drinking water. Small amounts of environmental BDCM are naturally occurring, formed from marine algae. BDCM has been used as a solvent for fats, waxes, and resins, as a flame retardant, as a heavy liquid for mineral and salt separation, and as a fire extinguisher fluid ingredient. The principal use of BDCM is as a chemical intermediate for organic synthesis and as a laboratory reagent.

BDCM evaporates readily, so that BDCM released from chemical facilities, waste sites, or drinking water enters the atmosphere as a gas. BDCM does not adsorb strongly to soils or sediments, nor does it tend to bioaccumulate.

BDCM is readily absorbed by inhalation and ingestion, and to a lesser extent through the skin. The most likely human exposure to BDCM is through drinking water.

Animal studies indicate that both the liver and kidney are susceptible to injury from BDCM exposure. This includes increased weight, inflammation, and/or degeneration, and decreased function. Rats and mice that were administered oral doses of BDCM displayed signs of central nervous system depression, including lethargy, labored breathing, and sedation (similar to other halogenated chemicals such as chloroform and carbon tetrachloride).

BDCM exposure has also been observed to result in developmental toxicity. Chronic oral studies indicate increased frequency of liver, large intestine, and kidney tumors in rats.

It is not known whether long-term exposures in humans contribute to the development of cancer. BDCM is classified as a Group B2, probable human carcinogen, by EPA, based on animal studies.

2-Butanone (Methyl Ethyl Ketone)

A colorless liquid with a sweet, but sharp odor, 2-butanone is volatile and will quickly evaporate into the air. The compound is manufactured in large amounts for use in paints, glues, and other finishes. It is also a natural product made by some trees, fruits, and vegetables.

The compound 2-butanone is often found dissolved in water or as a vapor in air. In water, it can be broken down into simpler chemicals by natural biological processes in about two weeks. In air, 2-butanone will break down under the influence of sunlight in one day or less. It does not adhere to soil appreciably. If it is spilled on soil, it can migrate through the soil either as a vapor or a liquid. It is relatively soluble in water which allows it to dissolve in and migrate with soil moisture and groundwater.

The most likely human exposure to 2-butanone is occupational. However, the general population could be exposed to 2-butanone in the air from drying paints and glue, and to 2-butanone in air, water or soil as a result of environmental contamination. Lastly, it has been determined that 2-butanone occurs naturally in some foods including: chicken, roasted nuts, cheese, fruit, dried beans, split peas, and lentils.

Greater than 50% of the inhaled 2-butanone is adsorbed onto the body. It is unknown how much of the oral and dermal doses are adsorbed. Humans can smell 2-butanone at levels below those that can cause adverse effects. At levels slightly above the odor threshold, people have reported mild irritation to the nose, throat, eyes, and skin.

In general, there are only observable effects at high doses in animal studies. Health effects resulting from inhalation or ingestion of 2-butanone include: respiratory irritation, kidney and liver abnormalities, underdeveloped offspring, and unconsciousness and death at high doses. Toxic effects to offspring is the EPA's critical or most sensitive effect. There are reports of behavioral effects in mice and baboons at low doses.

Very little long-term exposure data exist for 2-butanone in animals and humans. It is unknown whether 2-butanone causes cancer in animals or humans.

Carbon Tetrachloride

Carbon tetrachloride is a man-made clear liquid that evaporates very easily. It has been used in the production of the chlorofluorocarbons, in refrigeration fluids, and in

propellants for aerosol cans. In the past, carbon tetrachloride was widely used as a cleaning fluid especially in dry cleaning.

Most carbon tetrachloride released into surface soil or surface water will rapidly evaporate into the air. Carbon tetrachloride dissolves in water and can contaminate ground-water. Carbon tetrachloride can biodegrade fairly rapidly (6 to 12 months) into other chemicals.

Exposures to high concentrations of carbon tetrachloride can cause Central Nervous System (CNS) effects. The immediate symptoms include signs of intoxication, including headache and dizziness, along with nausea and vomiting. Prolonged exposures may cause coma or even death. Carbon tetrachloride also causes liver and kidney effects. The liver becomes swollen and tender, and fat tends to accumulate in the tissue, and, if exposure continues, it can lead to decreases in liver function. The formation of liver lesions is the critical or most sensitive toxic effect of carbon tetrachloride and serves as the basis of EPA's RfD. The kidneys produce less urine causing a buildup of fluids in the body, especially the lungs, and a buildup of waste products. Kidney failure is often the main cause of death in people overexposed to carbon tetrachloride.

Animal studies indicate that carbon tetrachloride does not cause birth defects but might decrease the survival rate of newborn animals. Also, animal studies have shown that oral exposure to carbon tetrachloride has increased the frequency of liver tumors. No studies have been performed on the inhalation route of exposure and frequency of liver tumors.

It is not known whether long-term exposures in humans contribute to the development of cancer. Carbon tetrachloride is classified as a Group B2, probable human carcinogen, by EPA based on animal studies.

Chloroform

Chloroform, also known as trichloromethane, is a colorless liquid with a pleasant, non-irritating odor and a slightly sweet taste. Chloroform is used primarily to synthesize other chemicals. Most chloroform found in the environment comes from chemical manufacturing plants, pulp and paper mills, chlorinated drinking water supplies, and chlorination of wastewater from sewage treatment plants. Chloroform is highly soluble in water, and it readily evaporates into air where it is ultimately degraded by photochemical

reactions. The most likely source of exposure to chloroform is through drinking water and/or breathing air containing chloroform contamination. It can also be absorbed through the skin. Inside the body, chloroform can be transported throughout the body, concentrating mainly in fat tissue, brain, liver, and kidney.

In humans, chloroform has been found to adversely affect the CNS, liver, kidneys, digestive system, heart, and circulatory system after exposure through inhalation or ingestion. CNS effects associated with human exposure to chloroform include dizziness, vertigo, headache, and in some cases death. When used as an anesthetic in the past, chloroform caused irregular heartbeat and low blood pressure. Anesthetic use was discontinued because of liver and kidney damage. Long-term exposure to low concentrations of chloroform also causes liver and kidney damage in humans. The formation of fatty cysts in the liver is the critical effect of chloroform in animals and serves as the basis of EPA's RfD.

In long-term animal studies, chloroform-induced liver and kidney damage has also been noted. Reproductive effects in mice associated with chloroform inhalation exposure include decreased ability to maintain pregnancy and an increase in birth defects. It is not known whether chloroform can cause similar reproductive effects in humans.

Although it is unknown whether long-term exposure contributes to the development of cancer in humans, liver and kidney tumors have been associated with oral exposure in mice and rats. Chloroform is classified as a Group B2, probable human carcinogen, by EPA based on animal studies.

1,1-Dichloroethane (1,1-DCA)

The compound 1,1-DCA is a man-made liquid chemical that is used industrially as a solvent and in the manufacture of other chemicals. When 1,1-DCA is released to surface water or surface soil, the chemical will evaporate into air. Although its water solubility is low, 1,1-DCA can migrate from soil into groundwater. Some 1,1-DCA found in the environment is a breakdown product of 1,1,1-trichloroethane. Human exposure to 1,1-DCA can result from breathing contaminated air or eating or drinking contaminated food or water.

Relatively little information is available on the health effects of 1,1-DCA in humans or animals. Although once used as a surgical anesthetic gas, the use of 1,1-DCA was discontinued when it was discovered that irregular heartbeats were induced at anesthetic

doses. Exposure to high levels of 1,1-DCA in air has caused death in animals. Long-term exposure to high levels of 1,1-DCA has caused kidney damage in laboratory animals. In addition, exposure of pregnant rats to 1,1-DCA in air resulted in delayed development in the offspring.

One laboratory study suggests 1,1-DCA may cause increased tumors in rats and mice, but the results are inconclusive. There is no evidence of similar harmful health effects in humans. In light of the results of animal studies, EPA has classified 1,1-DCA as a Group C possible human carcinogen.

1,1-Dichloroethene (1,1-DCE)

The compound 1,1-DCE is a man-made chemical that does not occur naturally in the environment. It is a clear, colorless liquid that has a mild, sweet, chloroform-like odor, and is used to make plastic products such as plastic wrap and flame-retardant fabrics.

The compound usually enters the body through inhalation and/or ingestion. It may also enter the body through the skin. The human health effects resulting from exposure to 1,1-DCE are unknown. In animal studies, brief exposures to high concentrations of 1,1-DCE have caused liver, kidney, heart, and lung damage, nervous system disturbances, and death. Prolonged exposure to lower concentrations of 1,1-DCE has also produced liver damage. This liver damage is the critical or most sensitive effect of 1,1-DCE and serves as the basis of EPA's RfD.

An increased risk of cancer was observed in animals exposed to 1,1-DCE, as was an increased risk of birth defects in the offspring of exposed pregnant animals. Based upon animal studies, 1,1-DCE is classified as a Group C possible human carcinogen.

1,2-Dichloroethene (1,2-DCE)

The compound 1,2-DCE is a man-made flammable liquid with a sharp, harsh odor. It is primarily used in the production of solvents and as an additive to dyes, lacquer solutions, perfumes, and thermoplastics. There are two forms of 1,2-DCE: cis-1,2-DCE and trans-1,2-DCE, which may occur separately or as a mixture.

In the environment, 1,2-DCE evaporates rapidly. When 1,2-DCE is released to either surface soil or surface water, almost all of the chemical will evaporate into air. When

1,2-DCE occurs underground, such as in landfills and chemical waste sites, it can dissolve in water and migrate with groundwater. In groundwater, 1,2-DCE breaks down to vinyl chloride, which ultimately breaks down to water, carbon dioxide, and chloride ions. Vinyl chloride, the initial breakdown product, is more toxic than 1,2-DCE.

This compound can enter the body by drinking water, eating food, or breathing air that contains 1,2-DCE. Because 1,2-DCE evaporates readily, inhalation is the most likely route of human exposure. Inhalation of high levels of 1,2-DCE can cause nausea, drowsiness, dizziness, and may result in death. Liver, heart, and lung damage were observed in laboratory animals after short- or long-term exposure to 1,2-DCE in air. Liver and lung damage was reported in animals that were fed 1,2-DCE. Death can also occur in animals that ingest large amounts of 1,2-DCE. Changes in blood chemistry is the critical or most sensitive effect and serves as the basis of the RfD used in the risk assessment.

The long-term health effects resulting from exposure to 1,2-DCE are unknown. Increased risk of cancer has not been reported in humans or animals exposed to 1,2-DCE.

Ethylbenzene

Ethylbenzene is a colorless liquid with a gasoline-like odor. Ethylbenzene occurs naturally in coal tar and petroleum, and is found in many synthetic products, including paints, inks, and insecticides. Gasoline contains approximately 2% ethylbenzene by weight.

Ethylbenzene evaporates easily into the air from soil or water. People living in urban areas or near factories or highways may be exposed to ethylbenzene in the air. Indoor air, on average, contains more ethylbenzene than outside air due to buildup from household products such as cleaning products and paints. Tobacco smoke also contains ethylbenzene.

Ethylbenzene can potentially enter the body through inhalation of vapors, through dermal contact with gasoline, paint vapors, or glue vapors, or through ingestion of food or water containing its residues.

Humans exposed to high levels of ethylbenzene have exhibited signs of dizziness and lethargy. Low-level exposure has been associated with eye and throat irritation. No deaths have been reported in humans exposed to ethylbenzene.

Short-term exposure to high concentrations of ethylbenzene in air is associated with liver, kidney, and nervous system damage, and death in laboratory animals. However, the

meaning of these effects are uncertain because of conflicting results and design weaknesses in many of the studies.

No data are available on long-term health effects in humans exposed to ethylbenzene. One long-term study showed increased tumors in rats treated with ethylbenzene, but the study was flawed. EPA has placed ethylbenzene in Group D, not classified for carcinogenicity, because of limited laboratory data and lack of adequate human data.

Methyl Chloride (Chloromethane)

Methyl chloride, which is a gas or pressurized liquid at ambient temperature, is used in the production of silicones, agricultural chemicals, synthetic rubber, and tetraethyl lead. It can be formed during the chlorination of drinking water. Methyl chloride is released in tobacco smoke and turbine exhaust. It is produced in nature by several photosynthesizing marine organisms, and is released through both brush and forest fires, and volcanoes.

Exposure to high concentrations of methyl chloride can result in dizziness, blurred vision, confusion, coordination problems, abdominal pains, nausea, and vomiting. These symptoms can be followed by delirium, convulsions, coma, and possibly death. Acute exposure to methyl chloride may also result in degenerative changes in the heart, liver, bone marrow, and kidneys. Dermal contact with liquid methyl chloride can result in anesthesia and possibly frostbite of the exposed tissue.

Chronic exposure to low concentrations of methyl chloride can cause CNS damage, and, in some cases, damage to the liver, kidneys, bone marrow, or cardiovascular system.

There are no adequate data available to suggest that methyl chloride is carcinogenic.

Methylene Chloride (Dichloromethane, MC)

MC is a man-made liquid chemical that is widely used as an industrial solvent and as a paint stripper. Because MC evaporates easily, it is released into the environment mainly in air, where it is broken down by sunlight. MC released to water or soil tends to volatilize to air, but may migrate to groundwater. MC is formed during water chlorination, and small amounts of MC may be found in some public drinking water supplies.

Absorption into the body occurs readily following exposure by breathing vapors or accidental ingestion. Occupational worker exposure to high levels of MC in air has resulted

in drowsiness, fatigue, lack of appetite, and light-headedness. Other effects include impaired reaction time and coordination, numbness or tingling of fingers and toes, and intoxication. The critical, or most sensitive effect, of MC exposure is liver damage observed in rats exposed to MC.

Chronic exposure of laboratory animals to high concentrations of MC by inhalation has resulted in an increased incidence of liver and lung cancer in mice and rats. MC has not been shown to cause cancer in occupationally exposed humans. Based on results from animal studies, EPA has classified MC as a Group B2, probable human carcinogen.

Methyl Isobutyl Ketone (4-methyl-2-pentanone)

Methyl isobutyl ketone (MIBK) is a man-made, colorless, transparent liquid with a camphor-like odor, which is used in a wide variety of industrial applications. The release of MIBK to the environment is mainly due to emissions and effluent from industries and vehicle exhaust, and from land disposal and ocean dumping of consumer products and industrial waste containing MIBK. MIBK is used and produced by manufacturers of coatings (i.e., lacquers, varnishes, paints), pharmaceuticals, pesticides, rubber processing chemicals, adhesives and by rare metal extractors. MIBK is moderately volatile and slightly soluble in water. It can be broken down by direct photolysis, by reaction with hydroxyl radicals in the atmosphere and by aerobic biodegradation in soil and water.

The most likely routes of human exposure to MIBK are inhalation and dermal contact while using products containing MIBK. People living in source areas may be exposed to MIBK by inhalation of contaminated air and ingestion of contaminated drinking water.

Occupational exposure to MIBK vapors at concentrations up to 500 ppm for 20 to 30 minutes a day has caused weakness, loss of appetite, headache, burning eyes, stomach ache, nausea, and sore throat. A few workers experienced insomnia, somnolence, heartburn, and intestinal pain. Several workers were found to have slightly enlarged livers and others complained of colitis.

Oral exposure to MIBK has caused kidney and liver effects in rats. In the past, these were the critical or most sensitive effects of MIBK and serve as the basis of the EPA's RfD. However, the oral RfD has since been withdrawn. There is no carcinogenicity information available for MIBK.

Tetrachlorethene (Perchloroethene, PCE)

PCE is a man-made chemical that is widely used for dry cleaning fabrics, for metal-degreasing operations, and in the manufacture of other chemicals.

Most PCE released into surface soil or surface water will rapidly evaporate into the air. PCE dissolves in water and can contaminate groundwater. PCE biodegrades slowly in groundwater. Some of the breakdown products of PCE are also hazardous chemicals such as trichloroethene (TCE) and vinyl chloride. Detectable background levels of PCE frequently occur in air and less frequently occur in drinking water.

Humans may be exposed to PCE by breathing air or ingesting food or water that has been contaminated with this chemical. For the general public, inhalation is the most likely route of exposure. PCE can occur in many household products including: suede protectors, water repellents, spot removers, and wood cleaners. Exposure to high concentrations in air, particularly in confined areas, can cause CNS effects, which may be expressed as dizziness, headache, sleepiness, confusion, nausea, and possibly unconsciousness and death.

Animal studies, conducted with concentrations much higher than those usually encountered in the environment, suggest that PCE can cause liver and kidney damage, developmental effects on fetuses, and toxicity to pregnant animals. Liver toxicity is the critical or most sensitive effect and serves as the basis for the RfD used in the risk assessment.

PCE causes liver cancer and kidney cancer in mice and rats. Studies of human PCE exposure and its relationship to increased cancer risk have produced conflicting results. Based on the evidence from animal studies, EPA once classified PCE as Group B2 probable human carcinogen. However, EPA has since withdrawn this classification and the carcinogenicity status is pending.

Toluene

Toluene is used as a solvent in the production of a variety of products and as a constituent in the formulation of gasoline and aviation fuels. Toluene can enter and affect the body if it is inhaled, it comes in contact with the eyes or skin, or if it is swallowed.

Exposure to toluene can cause many CNS effects. Toluene overexposure may cause fatigue, weakness, confusion, headache, dizziness, drowsiness, and irritation of the eyes,

respiratory tract, and skin. These symptoms have been reported in association with occupational exposure to airborne concentrations of toluene ranging from 50 ppm (189 mg/m³) to 1,500 ppm (5,660 mg/m³). Symptoms generally increase in severity with increased exposures.

Animal studies have shown that oral exposure to toluene has caused damage to both the liver and kidney. This damage is the critical effect of toluene and serves as the basis of EPA's RfD.

Toluene does not appear to cause cancer in animals or humans. No increased risk of cancer was detected in studies of occupationally exposed men. Similarly, toluene did not cause cancer in rats and mice exposed through inhalation.

1,1,1-Trichloroethane (1,1,1-TCA)

The compound 1,1,1-TCA is a man-made chemical that has many industrial and household uses. For example, it is used as a cleaning solvent to remove oil or grease from manufactured metal parts, and as a solvent to dissolve other substances such as glue and paint. Consumer products such as spot removers may contain 1,1,1-TCA. Much of the 1,1,1-TCA produced in the United States ends up in the atmosphere as a result of evaporation during use. 1,1,1-TCA released onto or into the ground can migrate into groundwater.

Inhalation is the major route of exposure in humans, but exposure can also occur through the consumption of contaminated food and water and by skin contact with 1,1,1-TCA-contaminated soil and water.

1,1,1-TCA is readily absorbed into the body following exposure by inhalation of air containing the vapor or ingestion of water or food containing 1,1,1-TCA. It also readily leaves the body with exhaled air.

Human inhalation of high levels of 1,1,1-TCA over a short period of time has resulted in CNS effects such as dizziness, light-headedness, and loss of balance and coordination. These health effects are readily reversible when exposure stops.

Studies in animals and humans have shown that mild liver effects result from long-term exposure. Kidney damage has also been reported in animal studies.

Chronic animal cancer studies were done on mice and rats dosed orally. No consistent pattern of an increased incidence of cancer was found, but the study was of limited

value because of the death of many of the animals. It is not known whether 1,1,1-TCA causes cancer in humans.

Trichloroethene (TCE)

TCE is a man-made chemical widely used as a cleaning agent and solvent for degreasing operations. Most TCE released into surface water or surficial soil will rapidly evaporate into the air. In the subsurface, TCE is moderately to highly mobile and can migrate to groundwater. TCE biodegrades very slowly in subsurface soils and groundwater. Microbial degradation products include dichloroethene and vinyl chloride.

Humans are most likely to be exposed to TCE in air. TCE also may occur in drinking water supplies and consumer products including metal cleaners, spot removers, rug cleaning fluids, paints, and paint removers. TCE may cause adverse health effects following exposure through inhalation, ingestion, or skin or eye contact. Exposure to high levels of TCE can cause CNS effects including drowsiness, dizziness, headache, blurred vision, lack of coordination, mental confusion, flushed skin, tremors, nausea, vomiting, fatigue, irregular heartbeat, and in some cases death. In the past, TCE was used as an anesthetic, but that use was discontinued when this substance was found to cause irregular heartbeats. Chronic exposure to TCE can cause liver damage and skin reactions, as well as CNS effects.

Exposure of laboratory animals to TCE has been associated with an increased incidence of a variety of tumors, including kidney, liver, and lung cancers. However, it is uncertain whether people exposed to TCE have a higher risk of cancer. In the past TCE was considered a Group B2 probable human carcinogen, this classification has been withdrawn by EPA, however, following further review.

Vinyl Chloride (VC)

VC, which is a gas or pressurized liquid at ambient temperature, is primarily used as a raw material in the chemical industry for the production of polymeric chemicals (e.g., polyvinyl chloride [PVC]) that are in turn used to manufacture a variety of plastic products. In addition, VC is a known degradation product of many chlorinated solvents including tetra-, tri-, and dichloroethenes. Most of the VC in the environment comes from the plastics industry's releases to air or water. In surface water or surface soil, VC evaporates readily.

Once in the air, VC breaks down rapidly to nonhazardous constituents. VC can migrate to groundwater and dissolve readily. Once in the groundwater, VC can persist for many years.

VC may cause adverse health effects following exposure by inhalation, ingestion, or by dermal or eye contact. VC inhalation can cause dizziness or sleepiness. VC was used as an anesthetic for a brief period of time. Breathing very high levels of VC can cause unconsciousness and in some cases death. On skin, exposure to liquid VC can cause burns. Noncarcinogenic effects associated with long-term occupational VC over exposure include hepatitis-like changes in the liver, changes in the immune system, and nerve damage.

VC has been shown to cause liver and lung cancer in rats, and liver cancer in workers occupationally exposed to air concentrations in the range of 25 ppm to greater than 200 ppm. Based on this evidence, EPA has classified VC as a Group A, a human carcinogen. Air standards as low as 1 ppm are specified for occupational exposure to VC in many countries.

Xylenes

Xylenes are components of coal tar and petroleum; however, the majority of xylenes used commercially are man-made. There are three isomers of xylene (ortho-, meta-, and para-xylene), which can occur separately or as a mixture, and are herein referred to as xylenes. Xylenes are used in solvent mixtures and cleaning agents, and are components of gasoline and other fuels.

Xylene evaporates easily and its presence is widespread in the environment. Xylene is released from industrial sources, automobile exhaust, operations employing it as a solvent, and chemical waste disposal sites. Xylenes can be detected in air in cities and industrial areas, and in some public drinking water supplies.

Exposure to xylene may occur by breathing xylene vapors, or eating or drinking xylene-contaminated food or water. Xylene is rapidly absorbed following inhalation or ingestion. Short-term human exposure to high levels of xylene in air (100 ppm to 230 ppm) causes irritation of the skin, eyes, nose, and throat, increased reaction time to a visual stimulus, impaired memory, stomach discomfort, and possible changes in the liver and kidneys. Xylene may be fatal if large enough concentrations are inhaled or ingested. There are no studies regarding the long-term effects of inhalation or ingestion of xylene by humans.

Exposure of laboratory animals to high levels of xylene in air resulted in changes in the cardiovascular system, changes in liver weights, and hearing loss. Decreased body weight and increased numbers of birth defects in unborn rats were observed when the mothers were exposed to high concentrations of xylene. The effects of long-term exposure to low concentrations of xylene have not been well studied in animals. The critical or most sensitive effect is a decreased body weight in laboratory animals.

Oral exposure to xylenes did not cause increased cancer in rats or mice. There are no human carcinogenicity data. EPA has placed xylene in Group D, not classified for carcinogenicity.

6.4.3 Quantitative Indices of Toxicity

Quantitative indices of toxicity were compiled for the dose-response assessment to be used in estimating the relationship between the extent of exposure to a contaminant and the potentially increased likelihood and/or severity of adverse effects. The methods for deriving indices of toxicity and estimating potential adverse effects are presented below. The indices of toxicity for the chemicals of potential concern are presented at the end of this section.

6.4.3.1 Categorization of Chemicals as Carcinogens or Noncarcinogens

For the purpose of this risk assessment, chemicals of potential concern were classified into two groups: potential carcinogens and noncarcinogens. The risks posed by these two types of compounds are assessed differently because noncarcinogens generally exhibit a threshold dose below which no adverse effects occur, while no such threshold has been shown to exist for carcinogens.

As used here, the term "carcinogen" means any chemical for which there is sufficient evidence that exposure may result in continuing uncontrolled cell division (cancer) in humans and/or animals. Conversely, the term "noncarcinogen" means any chemical for which the carcinogenic evidence is negative or insufficient. These classifications are dynamic; chemicals may be reclassified at any time that additional evidence becomes available that shifts the weight-of-evidence one way or the other.

Chemicals of potential concern have been classified as carcinogens or noncarcinogens based on weight-of-evidence criteria contained in the EPA's "Guidelines for Carcinogen Risk

Assessment" (1986b). Table 6-18 summarizes the five EPA weight-of-evidence categories. According to these EPA guidelines, chemicals in the first two Groups—A and B (B1 and B2)—are considered human carcinogens or probable human carcinogens based on sufficient evidence and should be subject to nonthreshold carcinogenic risk estimation procedures. Depending upon the quality of the data, Group C chemicals may also be subjected to these procedures. The remaining chemicals—in Groups D and E—are defined as noncarcinogens and should be subjected to threshold-based toxicological risk estimation procedures.

Exposure to some chemicals may result in both carcinogenic and noncarcinogenic effects. In these cases, both the carcinogenic and noncarcinogenic effects were evaluated and considered in the risk assessment process.

6.4.3.2 Assessment of Cardiogenesis

In contrast to noncarcinogenic effects, for which thresholds are thought to exist, scientists generally have been unable to demonstrate experimentally a threshold for carcinogenic effects. This has led to the assumption by federal regulatory agencies (e.g., EPA, Food and Drug Administration [FDA], and Occupational Safety and Health Administration [OSHA]) that any exposure to a carcinogen theoretically entails some finite risk of cancer. Depending on the potency of a specific carcinogen and the level of exposure, however, such a risk could be infinitesimally small.

Scientists have developed several mathematical models to estimate low-dose carcinogenic risks from observed high-dose risks. Consistent with current theories of carcinogenesis, EPA has selected the linearized multistage model based on prudent public health policy (EPA 1986b). In addition to employing the linearized multistage model, EPA uses the 95 UCL for doses or concentrations in animal or human studies to estimate low-dose slope factors (SFs). By using these procedures, the regulatory agencies are unlikely to underestimate the actual SFs (formerly called carcinogenic potency factors) for humans.

Using SFs, lifetime excess cancer risks can be estimated by:

$$\text{Risk} = \sum LADL_i \times SF_i$$

where:

$LADI_i$ = Exposure route-specific lifetime average daily intake, and

SF_i = Route-specific slope factor.

Using the multistage model, the carcinogenic risks for the oral, dermal, and inhalation routes of exposure are calculated as follows:

$$\text{Risk} = LADI_o SF_o + LADI_d SF_d + LADI_i SF_i$$

where subscript "o" indicates the oral route, subscript "d" the dermal route, and subscript "i" the inhalation route. SFs for the COPCs for the oral and inhalation exposure routes are presented in Table 6-19. The EPA's weight-of-evidence classification for the chemical and the type of cancer that may be associated with exposure to the chemical are also included in Table 6-19.

6.4.3.3 Assessment of Noncarcinogens

Risks associated with noncarcinogenic effects (e.g., organ damage, immunological effects, birth defects, skin irritation) are usually assessed by comparing the estimated average exposure to the acceptable daily dose, now called the RfD by EPA. The RfD is selected by identifying the lowest reliable no observed adverse effect level (NOAEL) or lowest observed adverse effect level (LOAEL) in the scientific studies in which effects were observed, then applying a suitable uncertainty factor (usually ranging from 10 to 1,000) to allow for differences between the study conditions and the human exposure situation to which the RfD is to be applied. NOAELs and LOAELs can be derived from either human epidemiological studies or animal studies; however, they are usually based on laboratory experiments on animals in which relatively high doses are used. Consequently, uncertainty or safety factors are applied when deriving RfDs to compensate for data limitations inherent in the underlying experiments and for the lack of precision created by extrapolating from high doses in animals to lower doses in humans. The five uncertainty factors commonly used are summarized in Table 6-20. Modifying factors are additional adjustment factors based on professional judgment.

ment and are incorporated in order to compensate for factors other than the usual uncertainty adjustments.

RfDs are generally calculated using the formula:

$$\text{RfD (mg/kg-day)} = \frac{\text{NOAEL or LOAEL (mg/kg-day)}}{(\text{Uncertainty Factor}) \times (\text{Modifying Factor})}$$

The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of the daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a portion of the lifetime, in the case of a subchronic RfD, or during the entire lifetime, in the case of a chronic RfD. The RfD is used as a reference point for gauging the potential effects of other exposures. Usually, exposures that are less than the RfD are not likely to be associated with health risks. As the frequency of exposures exceeding the RfD increases and as the size of the excess increases, the probability increases that adverse health effects may be observed in a human population. Nonetheless, a clear distinction that would categorize all exposures below the RfD as "acceptable" (risk-free) and all exposures in excess of the RfD as "unacceptable" (causing adverse effects) cannot be made (HEAST 1992). Noncarcinogenic risks are usually assessed by calculating a hazard quotient, which is the ratio of the estimated exposure to the RfD as follows:

$$\text{HQ} = \frac{\text{CDI}}{\text{RfD}}$$

where:

HQ = Hazard Quotient.
CDI = Chronic Daily Intake (exposure), and
RfD = Reference Dose (acceptable daily intake).

Hazard quotients are commonly summed across exposure routes to give a hazard index. A hazard index greater than 1 indicates that adverse effects may be possible, whereas a value of less than 1 means that adverse effects would not be expected. The higher the hazard index is above 1, the more likely it is that adverse effects could occur.

EPA is in the process of developing subchronic RfDs based on potential noncarcinogenic effects associated with exposure durations ranging from a few weeks to seven years. Short-term exposures can occur when an activity resulting in exposure is performed for a limited period of time or when a chemical degrades or disperses to negligible concentrations within a short period of time. The hazard quotient for subchronic exposure is obtained by dividing the estimated subchronic daily intake (SDI) by the subchronic RfD.

Chronic and subchronic RfDs for the oral and inhalation exposure routes are presented in Table 6-21. Other entries in the table that have not been discussed previously are as follows: the confidence level indicates the degree of confidence that should be placed in the RfD value and is usually obtained from the Integrated Risk Information System (IRIS) entry for a chemical; the critical effect is the effect or target organ affected by the smallest dose of the chemical that produces any adverse effect and that serves as the basis for the RfD; the RfD source is the source or reference for the RfD. The preferred source is EPA's IRIS data base, which contains confirmed values reflecting the consensus judgment of the agency. The second choice is EPA's Health Effects Assessment Summary Tables (HEAST), which contain information taken from final documents prepared by the EPA Office of Health and Environmental Assessment. The RfD basis is the vehicle in which the chemical was administered or the medium of exposure in the study(ies) that served as the basis for the RfD.

6.4.3.4 Route-to-Route Extrapolation of Reference Doses and Slope Factors

Once substances have been absorbed through the oral or dermal routes, their distribution, metabolism, and elimination patterns (biokinetics) are usually similar. For this reason, and because dermal route RfDs and SFs are usually not available, oral route RfDs and SFs are commonly used to evaluate exposures to substances by both the oral and dermal routes. When this is done, the oral toxicity values are adjusted by dividing the oral SF or multiplying the oral RfD by the chemical's absorption efficiency. This accounts for differences in a chemical's administered dose (i.e., the oral dose) and its absorbed dose. Because the COPCs evaluated here are all volatile, an absorption efficiency of 100% was assumed.

Although inhalation route biokinetics differ more from oral route kinetics than do the dermal route kinetics, oral RfDs and Sfs may also be used to evaluate inhalation exposures

(except in the case of exposure-point effects) if inhalation route RfDs and SFs are not available, and vice versa. Extrapolation of toxicological indices from one route to another is inappropriate if the critical effect for either route is at the point of contact.

6.4.4 Uncertainties Related to the Toxicity Assessment

6.4.4.1 Introduction

In order to evaluate the meaning of any risk assessment, one must consider the uncertainties in the assumptions made, the potential impact of quantitative changes in those assumptions on the risk estimates, and the relevance of the findings to real-world exposures and risks. Due to the number of assumptions, data points, and calculations, a degree of uncertainty is necessarily associated with the numerical toxicity values in any risk assessment.

6.4.4.2 Evaluation of Carcinogenic Toxicity Assessment Assumptions

The COPCs have been evaluated by EPA using its weight-of-evidence carcinogenicity evaluation criteria and have been placed in Group A, human carcinogens, or Group B, probable human carcinogens, based on sufficient data in humans or on sufficient data in animals and insufficient data in humans, respectively (EPA 1986b).

Rodent bioassay and epidemiological studies, such as those performed for the COPCs, would require tens of thousands of animals or humans in order to determine whether a chemical may be carcinogenic at low doses. As the relationship between tumor location, time to appearance, and the proportion of animals with cancer determines the estimated carcinogenic SF, animal bioassay or human epidemiological data are not routinely sufficient for directly estimating SFs at low doses. Therefore, by necessity, agencies such as EPA use carcinogenic extrapolation models for estimating low-dose SFs. Based upon prudent public policy, these agencies assume that there is no threshold dose below which carcinogenic risks will not occur. This is equivalent to the assumption that every dose above zero, no matter how low, carries with it at least a small but finite risk of cancer. The models are based on the assumption that the dose-response relationship is linear at low doses. This is contrary to approaches used for other toxic effects, for which thresholds are assumed to exist.

The current model favored by EPA and certain other federal regulatory agencies is the linearized multistage model. The agency then uses the statistically derived upper 95th

percentile confidence bounds, rather than a maximum likelihood value for the SF. The agency has concluded, based on theoretical grounds consistent with human epidemiological and animal data, that cancer follows a series of discrete stages (i.e., initiation, promotion, and progression) that ultimately can result in the uncontrolled cell proliferation known as cancer. Consistent with this conclusion, the use of the linearized multistage model permits an estimation of SF that is not likely to be exceeded if the real slope could be measured. However, compelling scientific arguments can be made for several other extrapolative models, which, if used, could result in significantly reduced values for SFs, many times lower than those estimated using the linearized multistage model. The one-hit model, used to estimate risks due to exposures above the linear range of the multistage model, is one such model. Thus, most of the current EPA SFs calculated in this fashion represent upper-bound values based on animal data, which should not be interpreted as necessarily equivalent to actual human cancer potencies. It is this conservative value, nevertheless, that is used in this risk assessment on policy grounds for the protection of public health.

6.4.4.3 Evaluation of Noncarcinogenic Toxicity Assessment Assumptions

Key assumptions used in assessing the likelihood of noncarcinogenic effects are that threshold doses exist below which various noncarcinogenic effects do not occur and that the occurrence or absence of noncarcinogenic effects can be extrapolated between species and occasionally between routes of exposure and over varying exposure durations. The threshold assumption appears to be sound for most noncarcinogens based on reasonably good fits of experimental data to the usual dose-response curves.

The other assumptions generally appear to be true to varying degrees. The effects observed in one species or by one route of exposure may not occur in another species or by another route, or they may occur at a higher or lower dose due to differences in the bio-kinetics of a compound in different species or when exposure occurs by different routes. The uncertainty in these assumptions is taken into account in the development of RfDs through the use of safety or uncertainty factors. These factors reflect uncertainty associated with species-to-species extrapolation and include safety factors to protect sensitive individuals. In addition to uncertainty factors, a modifying factor is applied to reflect a qualitative professional assessment of additional uncertainties in the critical study and in the entire data base for the

chemical not explicitly addressed by the preceding uncertainty factors. The modifying factor ranges from greater than 0 to 10 with a default value of 1 (EPA 1989b).

The uncertainty factors and modifying factors used by EPA are conservative (health-protective) in nature in that they tend to overestimate the uncertainties so that the RfDs obtained are unlikely to be too high. Use of the resulting RfDs tends to overestimate the potential for noncarcinogenic effects occurring at a given exposure level.

6.4.4.4 Route-to-Route Extrapolation of Reference Doses and Slope Factors

Route-to-route extrapolation of RfDs and SFs adds an additional source of uncertainty to the risk estimates obtained through their use. Such extrapolation may result in either under- or overestimation of the true risks for the extrapolated route. Although this practice adds uncertainty to the risk assessment process, it appears to be preferable to omitting exposure to a chemical by a route for which no RfD or SF is available from the quantitative risk assessment, which would lead to underestimation of the overall risks posed by the chemical.

6.4.4.5 Summary of Toxicity Uncertainties

The basic uncertainties underlying the assessment of the toxicity of a chemical include:

- Uncertainties arising from the design, execution, or relevance of the scientific studies that form the basis of the assessment; and
- Uncertainties involved in extrapolating from the underlying scientific studies to the exposure situation being evaluated, including variable responses to chemical exposures within human and animal populations, between species, and between routes of exposure.

Risk calculations may also be underestimated since many contaminants do not have toxicity values (i.e., slope factors and RfDs). In most cases, this is because not enough research has been performed in order to develop a slope factor and/or an RfD for a particular contaminant. This may result in an underestimation of site risks.

These basic uncertainties could result in a toxicity estimate, based directly on the underlying studies, that either under- or overestimates the true toxicity of a chemical in the circumstances of interest.

The toxicity assessment process compensates for these basic uncertainties through the use of safety factors (uncertainty factors) and modifying factors when assessing noncarcinogens, and the use of the upper 95th percentile confidence limit from the linearized multistage model for the SF when assessing carcinogens. The use of the safety factors and the upper 95th percentile confidence limit in deriving the RfDs and SFs ensures that the toxicity values used in the risk estimation process are very unlikely to underestimate the true toxicity of a chemical.

6.5 RISK CHARACTERIZATION

6.5.1 Introduction

This section combines the information developed in the exposure and toxicity assessment sections to obtain estimates of the potential risks posed to human health by Conrail Site contaminants. The process by which this is done is also explained in this section.

Risks due to carcinogenic and noncarcinogenic contaminants are usually assessed as discussed in Sections 6.4.3.2 and 6.4.3.3. Potential carcinogenic risks are assessed by multiplying the estimated lifetime average daily intake (LADI) of a carcinogen by its estimated SF to obtain the estimated risk, expressed as the probability of that exposure resulting in an excess incidence of cancer (i.e., more cancers than would normally be expected in that population). The potential for adverse effects resulting from exposure to noncarcinogens is assessed by comparing the chronic daily intake (CDI) or subchronic daily intake (SDI) of a substance to its chronic or subchronic RfD. This comparison is performed by calculating the ratio of the estimated CDI or SDI to the corresponding RfD, which is called a hazard quotient or hazard index. If the hazard index is less than 1, no adverse effects would be expected; however, if the hazard index is greater than 1, adverse effects could be possible.

The excess cancer risk or the hazard quotient for exposure to each chemical by each route of exposure, exposure pathway, and exposure case (CT or RME) initially are estimated separately. The separate cancer risk estimates are then summed across chemicals and across

all exposure routes and pathways applicable to the same population to obtain the total excess cancer risk for individuals in that population. Hazard quotients for noncarcinogens are only additive for chemicals that produce the same type of adverse effect (such as liver damage) and should be kept separate if the effects are different. Hazard quotients for subchronic and chronic effects were separately summed across all chemicals, exposure routes, and pathways applicable to the same population to obtain hazard indices for that population. Separate hazard indices for different types of effects were calculated only if the overall hazard indices approached or exceeded 1. Separate hazard indices were not calculated if the bulk of a hazard index value was due to a single chemical.

6.5.2 Summary Discussion of the Risk Characterization

6.5.2.1 Characterization of Contamination Present at the Site

The three phases of the RI/FS were designed to characterize the nature, extent, and limits of groundwater and soil contamination on the site. Possible source areas were identified based on a review of past activities at the site and previous sampling activities. Potential source areas and migration pathways were then investigated using various field techniques and by collection and analysis of samples. In this way, the nature of the contamination was characterized and its extent, to the limits possible, was defined.

6.5.2.2 Risk Assessment Benchmarks

EPA has adopted the policy that acceptable exposures to known or suspected carcinogens are generally those that represent an excess upper-bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} . In addition, EPA uses the 10^{-6} risk level as the point of departure for determining remediation goals for National Priorities List (NPL) sites.

For systemic toxicants (noncarcinogens) EPA defines acceptable exposure levels as those to which the human population, including sensitive subgroups, may be exposed without adverse effects during a lifetime or part of a lifetime, incorporating an adequate margin of safety (EPA 1989b). This acceptable exposure level is best approximated by a hazard index of 1. If the hazard index is less than 1, adverse effects usually would not be expected. However, as the hazard index increases beyond 1, the possibility of adverse effects occurring also increases.

6.5.2.3 Magnitude and Sources of Risks Posed by Site Contaminants

Tables containing estimates of exposures and associated risks for the scenarios described earlier in Section 6.3.3 are located in Appendix H and are summarized in Tables 6-22 and 6-23, respectively. Risk and hazard index estimates corresponding to the CT and RME cases are provided. According to EPA guidance (EPA 1989b), risk management decisions should be based on risk estimates for the RME case in order to ensure that the decisions are adequately protective of the public health. The CT case estimates are included to provide a measure of the central tendency of the potential risks and the magnitude of the uncertainty in the exposure assessment process.

Under existing site conditions, the highest estimated excess cancer risk (9.00×10^{-3}) was associated with residential groundwater usage in Plume 1 for the RME case. The bulk of this estimated risk, 42%, was due to ingestion of carbon tetrachloride in drinking water, and 28%, to inhalation of carbon tetrachloride and trichloroethene in shower vapors. The total contribution to the excess cancer risks in the groundwater usage pathway due to carbon tetrachloride and trichloroethene is 61% and 30%, respectively. Minor contributors to the pathways are 1,1-dichloroethene (5%), vinyl chloride (2%), and chloroform (2%).

The RME hazard index for the Plume 1 groundwater pathway is 201. Carbon tetrachloride accounts for 96% of the risk for the adult RME case, with 48% of the total risk due to ingestion of contaminated water and 48% due to inhalation of contaminated water. No RfD or RfC values have been established for potential noncancer effects of TCE. Therefore, TCE could not be included in the quantitative assessment of the potential non-cancer effects of site related contaminants.

The inhalation of indoor airborne contaminants in Plume 1 areas also presented a high estimated excess cancer risk (1.71×10^{-4}) for the RME case. The excess cancer risks for the adult RME case are primarily due to carbon tetrachloride (93%), trichloroethene (2%), and 1,1-dichloroethene (5%). The hazard index for this pathway is 10, indicating that there is a potential for adverse noncarcinogenic risks to these individuals.

The exposure-point concentrations used for estimating the groundwater usage risks are listed in Appendix H. The exposure-point concentrations of carbon tetrachloride (2.48 mg/L), chloroform (0.148 mg/L), trichloroethene (13.0 mg/L), and 1,1-dichloroethene (0.048 mg/L), were well above their primary drinking water standards of 0.005 mg/L, (0.1 mg/L [as total

trihalomethanes]) 0.005 mg/L, and 0.007 mg/L, respectively (EPA 1991b). The groundwater exposure-point concentration for vinyl chloride (0.0065 mg/L) is also above its primary drinking water standard (0.002 mg/L).

Based on the risk estimates discussed above, residents of homes in areas located above Plume 1 (i.e., Charles Avenue area, County Road 1 area, and Vistula Avenue areas) could potentially be affected by groundwater contamination directly through groundwater usage, and indirectly through volatilization of contaminants from the groundwater to indoor air in their homes. However, both of the inhalation routes (shower vapor inhalation and indoor air inhalation) are based on mathematical models that use conservative (i.e., health-protective) assumptions. The risk estimates presented for the two inhalation pathways may be overly conservative and may overestimate the risks associated with exposures through these pathways.

The Plume 2 excess cancer risks also exceed the upper end of the range of risks deemed acceptable by EPA. The groundwater usage RME risk is 1.45×10^{-3} , while the RME risk from the indoor air inhalation pathway is 1.11×10^{-5} . The bulk of the RME risk for the groundwater pathway is due to benzene inhalation (90%), while inhalation of carbon tetrachloride accounts for 95% of the risks from exposure to indoor air. However, carbon tetrachloride accounts for some of the cancer risks associated with the Plume 2 groundwater usage, while chloroform and trichloroethene account for 2% and 3%, respectively, of the indoor air inhalation pathway. The exposure-point groundwater concentrations used in the Plume 2 risk estimate calculations for both carbon tetrachloride (0.076 mg/L) and trichloroethene (0.01 mg/L) exceed the primary drinking water standard for both compounds, which is 0.005 mg/L (EPA 1991b).

The hazard index for the RME case of the Plume 2 groundwater usage pathway is 4.16. This indicates that a potential for adverse noncarcinogenic risks to the adult population may exist. The RME hazard index for the indoor air pathway, is less than 1.0, indicating that there are no apparent adverse noncarcinogenic risks from exposures to this pathway.

The current worker exposure through inhalation of airborne contaminants on the Conrail facility also poses a potential excess cancer risk (3.40×10^{-3}). Carbon tetrachloride accounts for 97% of this risk, and vinyl chloride accounts for 2.5% of the risk. The RME hazard index for this pathway is 253, with carbon tetrachloride accounting for 100% of this

risk. The exposure-point concentrations for this pathway are presented in Table H-17. The model used to derive the ambient air exposure-point concentrations incorporates conservative assumptions, and may result in overestimating the potential risk to Conrail railyard employees.

Future scenarios evaluated in this risk assessment are limited to worker exposures, since future residential exposures are expected to remain the same as the current scenarios. The RME cancer risk to workers in a future excavation scenario is 8.50×10^{-5} for the soil contact pathway. Trichloroethene and vinyl chloride account for 81% and 14% of the risks, respectively. The noncarcinogenic risk to workers via this pathway is 48.1. Acetone (6%), carbon tetrachloride (5%), 1,2-DCE (67%), ethylbenzene (6%), and 4-methyl-2-pentanone (16%) account for this risk. Both the cancer and noncarcinogenic risks to workers from the inhalation exposure pathway in a future excavation scenario are less than the EPA limits. It should, however, be reemphasized that only VOCs were evaluated. No inference can be made concerning the risks from exposure to non-VOC compounds.

6.5.2.4 Nature of Potential Adverse Health Effects

The chemicals most responsible for potential cancer risks from residential Plume 1, residential Plume 2, and facility-related worker exposures are carbon tetrachloride and trichloroethene. Benzene, vinyl chloride, chloroform, and 1,1-dichloroethene also contribute to the total risk. EPA has classified benzene and vinyl chloride as Group A, human carcinogens, based on epidemiological studies. Benzene affects blood formation and the immune system. Vinyl chloride has caused lung cancer in occupationally exposed workers. Carbon tetrachloride, trichloroethene, and chloroform have been classified as Group B2, probable human carcinogens, based on carcinogenicity in animals. The compound 1,1-dichloroethene is classified as a group C carcinogen; there is limited evidence that it is carcinogenic to animals and, therefore, it is considered a possible human carcinogen.

Site contaminants that could pose potentially significant noncarcinogenic health effects include carbon tetrachloride, 1,1-dichloroethene, and 1,2-dichloroethene. In humans, exposure to carbon tetrachloride may cause nausea, liver necrosis or cirrhosis, kidney problems, and marked depression of the central nervous system. Humans exposed to 1,1-dichloroethene have experienced hepatotoxicity and kidney dysfunction leading to

histopathological changes in the kidney. Humans exposed to 1,2-dichloroethene have experienced decreased red blood cell counts.

6.5.2.5 Level of Confidence/Uncertainty in the Risk Estimates

These matters are discussed fully in earlier sections of this report. Briefly, the level of confidence in the exposure estimates is moderate to good. The level of confidence in the toxicity estimates varies from chemical to chemical as shown in Tables 6-19 and 6-21.

Overall, the level of confidence in the risk estimates is also moderate to good. As noted earlier, however, the nature of the risk assessment process strongly favors overestimation of the true risks.

6.5.2.6 Major Factors Driving Risks

There are several exposure pathways and chemicals driving the estimated potentially significant risks for the Conrail Site. These risks are primarily due to chemicals present at elevated levels that appear to be associated with past railyard activities.

Those risks to residents living in both Plume 1 and Plume 2 residential areas under existing conditions that exceed EPA's benchmark levels of 10^{-6} for carcinogens and 1.0 for noncarcinogens are related to chemicals released during past railyard activities that have migrated north of the facility in groundwater. Exposure pathways contributing to these potential risks include: ingestion of drinking water, dermal contact with water while showering, inhalation of vapors while showering, and inhalation of indoor airborne contaminants. Other potentially significant risks estimated under existing conditions are risks posed to railyard workers through inhalation of airborne contaminants volatilized from subsurface sources.

The highest estimated potential cancer risk under existing conditions was associated with Plume 1 residential exposures. The bulk of this risk is due to ingestion and inhalation of carbon tetrachloride, chloroform, trichloroethene, 1,1-DCE, and vinyl chloride. Plume 2 residential exposures were also high, with the bulk of the risk due to ingestion of carbon tetrachloride and inhalation of carbon tetrachloride, chloroform, and benzene. Worker exposures were evaluated for one pathway, inhalation; the risk associated with this pathway is due primarily to carbon tetrachloride with a smaller contribution from vinyl chloride.

Hypothetical future worker scenarios involving excavations on the Conrail facility revealed noncancer risks via the soil contact pathway.

6.5.3 Risk Characterization Uncertainties

The risk characterization combines and integrates the information developed in the exposure and toxicity assessments; therefore, uncertainties associated with these assessments also affect the degree of confidence that can be placed in risk characterization results. Sections 6.3.4 and 6.4.4 provide full discussions of the factors causing uncertainty in the exposure and toxicity assessments, respectively.

For the exposure assessment, factors that would likely cause overestimation of the true exposures were:

- The directed nature of the sampling program;
- The use of the upper 95th percent confidence limit on the mean or the maximum observed value for the source concentrations;
- The use of many 90th percentile values in the exposure estimation calculations;
- The use of the steady-state assumption for source concentration estimates for all but the soil excavation (future) scenario; and
- The use of transport models, which incorporate their own conservative assumptions, to estimate exposure-point concentrations in ambient (facility) air and residential indoor (basement) air.

Factors that could lead to underestimation of the exposures are:

- The use of some sample quantitation limits that could have resulted in missing low concentrations of some chemicals that might pose significant risks; and
- Limiting this evaluation to VOC compounds only.

A factor that could have led to either overestimation or underestimation of the exposures is:

underestimating the true risk, yet not so conservative that the resulting risk estimate turns out to be unreasonably high. When in doubt, the risk assessor will usually elect to err in favor of protecting human health and select a value that results in overestimating the true risk. The nature of the risk estimation process itself virtually ensures that the true risks are much more likely to be overestimated than underestimated.

The last uncertainty factor to consider is the likelihood of the postulated exposures actually occurring. The exposure pathways identified as complete under current land use conditions are all plausible, and exposure is either presently occurring by these pathways or such exposure could reasonably be expected. The postulated frequencies of occurrence may overestimate routine occurrence but could certainly reflect the reasonable maximum occurrence.

Future land uses and potential exposure pathways are expected to remain largely the same as under current site conditions. A hypothetical future railyard excavation scenario was included in the risk assessment at EPA's request to evaluate potential risks to future railyard workers if subsurface contamination was unearthed during excavation or soil moving activities. This could occur if, in the future, underground cables, pipes, or tanks are installed, removed, or repaired.

6.6 ECOLOGICAL ASSESSMENT

This Baseline Ecological Assessment (EA) focuses on existing and potential risks posed by site-derived contaminants to nearby natural habitats and associated flora and fauna in the absence of remedial action. This EA supports the remedial decision-making process by estimating the magnitude of potential risk to the natural environment. This information can then be incorporated into decisions regarding whether remediation is required, and, if so, the extent of remediation.

The objective of this EA at the Conrail Site was to screen the surface waters and sediments of nearby aquatic and wetland habitats for site-related contaminants to estimate the potential risk that contaminants pose to the natural environment. Therefore, a screening-level risk assessment was conducted using available or readily obtainable data with comparisons based on established criteria and reasonable maximum exposure assumptions. The results of this assessment will be used to determine whether a detailed EA is warranted (possibly

including toxicity testing of field-collected media, additional chemical analysis of the media or biological tissues, or modeling (EPA 1991c)). The determination of whether a more detailed EA is warranted lies with the Risk Manager (i.e., EPA), and will be made based on the results of this screening-level EA.

The EA was conducted according to the current regional and national EPA guidelines specified in the following documents:

- *Guidance for Conducting Remedial Investigation and Feasibility Studies Under CERCLA: Interim Final* (EPA/540/G-89/004) (EPA 1989e);
- *Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation Manual* (EPA/540/1-89/001) (EPA 1989c);
- *Framework for Ecological Risk Assessment* (EPA/630/R-92/001) (EPA 1992d); and
- *Region V Scope of Work for Ecological Assessment* (EPA 1991c).

Following these guidelines, the EA consists of six subsections. Following this introductory section, Section 6.7 identifies the habitats and special ecological features of the site. In Section 6.8, contaminants considered to be a potential threat to the natural environment are identified as Chemicals of Potential Ecological Concern (CPECs), and the criteria used in the CPEC selection process are described. The exposure pathways and receiving natural habitats are characterized in Section 6.9 along with assessment endpoints, and Section 6.10 describes the significance of the selected CPECs. Section 6.11 synthesizes the preceding ecological information and presents conclusions. References used in this EA are provided in Section 8.

Information is provided in previous sections of this RI report on site contaminants, geology, hydrology, soils, and other characteristics. To avoid redundancy, reference is made to these sections when appropriate.

6.7 CHARACTERIZATION OF NATURAL FEATURES IN THE VICINITY OF THE SITE

6.7.1 Introduction

The purpose of this section is to identify and describe natural and special ecological features (e.g., endangered or threatened species) present in the vicinity of the Conrail Site. Information was obtained from federal and state agencies and from observations made during the ecological/sampling site visit, which was conducted in December 1992. The physical setting (geology, hydrogeology, and climate) of the habitats will not be addressed, since the physical setting for the entire area has been described in Sections 1 and 3. Likewise, the nature and extent of contamination in nearby natural habitats is presented in Section 4, and corresponding unabridged tables of chemical data are presented in Appendix E.

6.7.2 Characterization of Natural Habitats

The focus of the EA is on the risk to aquatic organisms from surface water and sediments and to terrestrial wildlife from the surface water from Baugo Bay, the St. Joseph River downstream of Crawford Ditch, and the three ponds on the southern boundary of the Conrail facility (see Figure 2-5). These areas were chosen because they have the greatest potential to be influenced by the migration of site contaminants. Based on the results of the three phases of field investigations, Baugo Bay may receive discharges of groundwater that is contaminated with CCl_4 and TCE. Also, based on these investigations, Crawford Ditch receives surface runoff via the drainage network on the Conrail facility. Crawford Ditch flows north from the facility and empties into the St. Joseph River. The on-site ponds receive surface runoff from the site as well as discharges from the Conrail drainage network.

6.7.2.1 Conrail Site

Elkhart and St. Joseph counties in Indiana lie within a transition area from forestland to tall grass prairies (Risser *et al.* 1981). The city of Elkhart in Elkhart County and Osceola and Mishawaka in St. Joseph County are heavily populated urban areas. Plant communities within the limits of such cities are isolated and rare. These isolated urban plant communities probably exist because their locations are unsuitable for human habitation due to moisture,

soils, or topography. Surrounding these three cities are agricultural zones with pockets of native vegetation.

To the north, east, and west of the Conrail Site lies a heavily developed commercial/residential zone. This commercial/residential zone, located within the Elkhart city limits, has been significantly altered by development (i.e., excavations and filling). Therefore, little, if any, native vegetation or natural soil horizons are likely to exist adjacent to the site. Instead, native flora and fauna have probably been displaced with species more tolerant of human disturbances and activities. To the south of the Conrail Site lie agricultural areas with pockets of residential developments. These farmlands can support a greater diversity of faunal and floral species.

The Indiana Department of Natural Resources (IDNR) and the United States Fish and Wildlife Service (USFWS) have been contacted with regard to threatened, endangered, or special-concern species and significant habitats within 2.0 miles of the site. The USFWS has indicated no federally listed threatened or endangered species exist near the site (Hudak 1993). IDNR has indicated that four threatened, endangered, or rare species are located within 2 miles of the site (Martin 1993). These species and their locations are listed in Table 6-24. No impacts are expected to the two rare plant species from contaminants migrating from the Conrail Site. Both sightings are historic, and the locations of the sightings have been significantly altered by development along the St. Joseph River. Also, no impacts to the badger are expected from the Conrail Site. The latest sighting was north of the St. Joseph River and upgradient of the site. In addition, the badger is primarily terrestrial and not expected to feed on aquatic resources in the site area. Two recent sightings of the Cooper's hawk suggest that it could potentially be impacted by contaminants migrating from the Conrail Site. The first sighting location overlaps the St. Joseph River downstream of Crawford Ditch. The second location is adjacent to the easternmost pond.

6.7.2.2 St. Joseph River

The portion of the St. Joseph River adjacent to the site is located approximately 0.5 miles upstream of the Twin Branch Dam. This dam has changed the river's characteristics and fish habitat. The river in the area of the dam has become a large reservoir and sediment trap with no noticeable flow. The average flow rate in the area of the river is 3,255 cubic

feet per second (United States Geological Survey [USGS] 1993). The average width of the river is 500 feet to 700 feet and the average depth is 8 feet (Ledet 1979). The substrate is primarily sand and silt, with a small amount of gravel (Ledet 1979). The river is intensively used for recreational purposes. Several boat ramps exist along this portion of the river.

The IDNR Division of Fish and Wildlife has conducted fish surveys in the St. Joseph River. Two of the sampling locations were located in the vicinity of the site. The first sample location was upstream from the Twin Branch Dam at the Bittersweet Road Bridge. This sample location is downstream of the confluence of Baugo Bay and the river. The second sample location was the confluence of the river and Cobus Creek to ½ mile upstream. This sample location is in the vicinity of the Phase III RI background sediment and surface water sample locations for the St. Joseph River (see Figure 2-5) (Ledet 1979, 1989). The results of the fish sampling effort are provided in Table 6-25.

A fish consumption advisory is in effect for carp (Group 2 Advisory) caught in the St. Joseph River in Elkhart and St. Joseph counties (Indiana State Department of Health [ISDH] 1992). A Group 2 Advisory indicates that adult men and women who are not of childbearing age should consume no more than one meal containing carp per week, while women of childbearing age should not consume any fish. A Group 2 Advisory indicates that the levels of Chlordane, PCBs, DDT, and Dieldrin exceed the Food and Drug Administration's action levels for tissue. These levels are 0.3, 2, 5, and 0.3 ppm, respectively.

6.7.2.3 Baugo Creek and Baugo Bay

Baugo Creek, which is located upgradient and east of the site, flows in a northerly direction until it crosses the Conrail tracks. This portion of the creek was used for the collection of background samples (see Figure 2-5). North of the Conrail tracks, the creek widens and forms Baugo Bay. Baugo Bay empties into the St. Joseph River near the Charles Avenue residential area, and lies downgradient of the site. The characteristics of Baugo Bay are similar to those of the St. Joseph River (see Section 6.7.2.2).

Baugo Bay, and the ponds). In contrast, numerous inorganics were detected. The results were compared to chronic ambient water quality criteria (AWQC) for the protection of aquatic life. Chemicals whose concentrations exceed the AWQC are listed in Table 6-27, and were chosen as CPECs. There are no surface water CPECs for Baugo Bay. The only CPEC for the St. Joseph River is lead. Lead and zinc are CPECs for the ponds area (see Table 6-24); the values that exceeded AWQCs were detected in samples collected from the westernmost pond.

6.8.2 Sediment

6.8.2.1 Organics

Several volatile organics were detected in the sediments collected from the three locations of concern as well as in the associated background samples. Toluene was detected in all areas, indicating an ephemeral toluene source may be present; however, the concentrations were slightly above detection limits. Also, toluene is rapidly dissipated in the environment as indicated by no detections in surface water. Singular detections of 2-butanone (MEK), CCl_4 , and dichloroethene (DCE), and two detections of trichloroethene (TCE) were found in the sediments collected from the St. Joseph River. This pattern of detections suggests no pervasive sediment contamination is present that would likely contribute a significant risk to aquatic life. In addition, these contaminants are volatile and would be expected to rapidly dissipate in open water. Finally, they were not detected in any surface water samples, except for TCE, which was detected once at a concentration below its AWQC.

The first step in the selection of sediment CPECs was the development of sediment quality criteria (SQC) for the compounds detected on site. To accomplish this, the equilibrium partitioning (EP) approach was used for non-polar organics in an attempt to model chemical movement from one medium to another (EPA 1988d). EP is based on the premise that the distribution of a contaminant in an environmental medium (i.e., sediment) is controlled by continuous exchange with other environmental media (i.e., water and biota). This distribution is in equilibrium or steady-state and is illustrated in Figure 6-3. EPA (1988d) and the Ontario Ministry of Environment (OME) (1991) have used this method to develop a limited number of SQC values, which are listed in Table 6-28.

For contaminants having no EPA- or OME-calculated interim criteria, SQC values were calculated using the following equation (DiToro *et al.* 1991):

$$SQC = f_{oc} \times K_{oc} \times FCV \text{ (or FRV)} \quad [\text{Eq. 6-1}]$$

where: SQC = Sediment quality criteria,
 f_{oc} = Organic fraction of the sediments (assumed to be 0.01),
 K_{oc} = Organic carbon partition coefficient,
FCV = Final chronic value, and
FRV = Final residual value for water.

The organic fraction or f_{oc} was assumed to be 1 %, which is a conservative estimate. The K_{oc} for each chemical was calculated from the octanol-water partition coefficient (K_{ow}) using the following equation (EPA 1988d):

$$\text{Log}_{10} (K_{oc}) = 0.00028 + 0.983 \times \text{Log}_{10} (K_{ow}) \quad [\text{Eq. 6-2}]$$

Calculated K_{oc} values were used rather than literature K_{oc} values because the experimental procedures for determining K_{ow} values are simpler than those used for K_{oc} values. Published K_{ow} values are probably more accurate than published K_{oc} values because interferences caused by dissolved organic carbon and particle effects do not have to be considered or accounted for in experiments designed to determine K_{ow} values (EPA 1988d).

The EP approach cannot be performed on compounds with log K_{ow} values less than 2.00, since these compounds are polar and readily dissolve in water. Therefore, this approach cannot be used on volatile organics.

The FCV or FRV value was derived according to EPA guidance (Stephan *et al.* 1985) by using the Genus Mean Acute Values (GMAVs) of the four most sensitive genera in order to calculate a final acute value (FAV). The FAV can then be divided by the acute-chronic ratio to yield the FCV. The calculated FCV values are listed in Table 6-28. The AQUIRE

(AQUatic toxicity information Retrieval) data base (EPA 1993) was used to obtain information on the most sensitive genera. If contaminants weren't available on AQUIRE (e.g., indeno (1,2,3-cd)pyrene and benzo(g,h,i)perylene), a similarly structured chemical with a conservative FCV value was substituted. Table 6-29 summarizes the SQCs calculated using the method described above for freshwater. Note that the chemicals for which SQC had to be calculated had only FCV, not FRV, values. Therefore, no FRV values are listed on Table 6-29.

As shown in Table 6-30, no organic CPECs were chosen for Baugo Bay or the St. Joseph River downstream of Crawford Ditch. One sample collected from the St. Joseph River downstream of Crawford Ditch exceeded the calculated criteria for 4-methylphenol. This contaminant, however, was present in the background sample at a concentration greater than the downstream sample. Therefore, it appears that the contamination from 4-methylphenol is not related to the site. For the ponds, five contaminants were considered as CPECs: 4,4'-DDT, 4,4'-DDD, 4-methylphenol, fluorene, and Aroclor 1254.

6.8.2.2 Inorganics

To evaluate the inorganic sediment contamination, the OME Provincial Sediment Quality Guidelines (PSQG) were used (OME 1991). The PSQG were developed for the Province of Ontario, Canada, and, therefore are not legally enforceable in the United States. However, since corresponding regulations for inorganic sediment quality are not available at this time in the United States, the PSQG values serve as useful guidelines for identifying potential CPECs. The PSQG values are provided in Table 6-31.

The inorganic results were compared to the conservative Lowest Observed Effect Level (LOEL). OME assumes that if the concentration is equal to or lower than the LOEL, then no significant ecological impact is expected. Therefore, if the results from sediment collected from the areas of concern were less than the LOEL, the inorganic contaminant would pose no significant risk to benthic organisms.

Because some of the background samples collected from Baugo Bay and the St. Joseph River exceeded established criteria, the data sets were compared to each other to determine whether a significant difference exists between the background results and the sampling results. A t-test was calculated using the background and sampling data sets. This

test assumes that the variable of interest has a normal distribution with a common variance. To check the variance, the F-test was calculated using the following equation:

$$F = s_1^2/s_2^2 \quad [\text{Eq. 6-3}]$$

where: s_1^2 = the square of the standard deviation of sample group 1, and
 s_2^2 = the square of the standard deviation of sample group 2.

The variance was statistically different if the calculated F value exceeded the tabulated F value with $df_1 = n_1 - 1$ and $df_2 = n_2 - 1$ (df = degree of freedom).

If there was a common variance, the t-test was calculated using the following formula independent of sample size:

$$t = \frac{(y_1 - y_2) - D_0}{s\sqrt{1/n_1 + 1/n_2}} \quad [\text{Eq. 6-4}]$$

where: y_1 = mean of sample group 1.
 y_2 = mean of sample group 2.
 D_0 = specified value (assume 0).

$$s = \sqrt{(n_1 - 1)s_1^2 + (n_2 - 1)s_2^2/n_1 + n_2 - 2}$$

n_1 = number of samples in data group 1, and
 n_2 = number of samples in data group 2.

If the absolute value of the calculated t value was greater than the tabular t value for the 95% confidence limit for $n - 1$ degrees of freedom, then the data sets were considered statistically different.

For samples that did not have a common variance (i.e., failed the F-test), the following equation was used:

$$t' = \frac{y_1 - y_2}{\sqrt{(s_1^2/n_1) + s_2^2/n_2}} \quad [\text{Eq. 6-5}]$$

and t' was rejected if:

$$|t'| > \frac{(t_1 s_1^2/n_1) + (t_2 s_2^2/n_2)}{(s_1^2/n_1) + (s_2^2/n_2)} \quad [\text{Eq. 6-6}]$$

Where t_1 and t_2 are the critical t-values based on $n - 1$ degrees of freedom.

The results of the t-test calculations for Baugo Bay and the St. Joseph River are provided in Tables 6-32 and 6-33, respectively.

Based on the results of the t-test, there was no significant difference between results from the background locations and results from the sampling locations for Baugo Bay and St. Joseph River. Therefore, the contamination does not appear to be site-related. For Baugo Bay, however, some of the mean concentrations were greater than two times the background mean concentrations. The risk posed by this difference cannot be fully attributed to the site; therefore, no inorganic CPECs were selected for these locations.

A t-test was not performed for the pond area because no background samples were available for comparison. The following inorganics are CPECs for the pond area: arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, and zinc.

6.9 EXPOSURE ASSESSMENT

Since no CPECs were determined for Baugo Bay or the St. Joseph River, the exposure assessment will focus on the on-site ponds.

6.9.1 Contaminant Release, Migration, and Fate

The Conrail railyard is the suspected source of contamination to the ponds. Two contaminant migration mechanisms are potentially operating at the facility, storm water runoff and groundwater.

Contaminants migrating through storm water runoff may be found in the water and sediment of the ponds. In this exposure scenario, contaminants could directly migrate from the facility through the drainage network. An outfall from the drainage network is located in the westernmost pond. Also, the contaminants of concern can be carried by suspended soil particles.

CCl_4 , TCE, and DCE can serve as organic carrier solvents in groundwater to aid in the migration of predominantly insoluble organics. The flow of groundwater beneath the site, however, is towards the St. Joseph River. The ponds are upgradient of the river. Therefore, the CPECs would be prone to bulk migration through the drainage network rather than migration by dissolution in followed by migration with groundwater.

6.9.2 Potential Exposure Pathways and Ecological Receptors

Under existing and future site conditions, four general categories of ecological receptors might be exposed to CPECs at the Conrail Site. Potentially exposed receptors include:

- Aquatic biota in the ponds;
- Semiaquatic wildlife and terrestrial wildlife that depend on the aquatic environment for a fraction of their food or habitat needs;
- Upland wildlife and birds that use the ponds as a water source; and
- Plants growing along or adjacent to the ponds.

The potential exposure pathways for these receptors vary among receptor types:

- Benthic invertebrates (especially burrowing taxa) and bottom-feeding fish are the primary receptors of contaminated sediments through dermal contact and absorption, ingestion, and, indirectly, by feeding on contaminated food.

- Other fish and aquatic invertebrates that inhabit the sediment surface and water column are primary receptors of water column contamination.
- Aquatic, semiaquatic, and terrestrial secondary consumers can be exposed to CPECs through food chain bioconcentration and bioaccumulation, and through direct contact and incidental ingestion of contaminated sediments and surface waters.
- Aquatic plants are receptors of sediment contamination through uptake from interstitial water.
- Plants growing on the site or adjacent to the water bodies could be exposed through uptake by roots.

The potential exposure pathways and ecological receptors listed above were evaluated for inclusion in the EA on the basis of affected media and CPECs. With the exception of lead, which is a CPEC in St. Joseph River water, all other CPECs were detected in the ponds and primarily in the sediments. Biota most likely to be affected by sediment contamination are the benthic organisms. Therefore, for the purposes of this EA, only sediment-dwelling organisms (benthos) will be evaluated for potential risks from exposures to CPECs in the sediment pathway. Upland terrestrial flora and fauna were excluded from this EA primarily due to the lack of any apparent off-site surface soil or off-site air contamination. This exclusion includes potential effects to the Cooper's hawk, a state special-concern species. Habitat requirements for the Cooper's hawk include a variety of different vegetative communities. The hawk prefers to nest in mature stands of trees, and males will not hunt within 0.5 miles of the nest (Brown and Amadon 1989). Cooper's hawk will travel up to 2 miles from the nest to hunt over open fields and along woodland margins (Brown and Amadon 1989). Food requirements for Cooper's hawk consist of a number of medium-sized vertebrates including chickens as well as some gamebirds (i.e., quail, grouse, and doves). Other favored prey include robins, starlings, meadowlarks, flickers, and blackbirds. The Cooper's hawk will also feed on chipmunks, squirrels, young rabbits, and bats (Audubon Society of New Hampshire [undated]). Since the food requirements and nesting requirements were not connected to the water bodies of concern, a pathway involving the Cooper's hawk was eliminated.

6.9.3 Assessment Endpoints

EPA guidance describes ecological endpoints of two general types: assessment endpoints and measurement endpoints. Assessment endpoints are "formal expressions of the environmental values that are to be protected" from impacts of site contaminants. Measurement endpoints are "measurable environmental characteristic(s)... related to the valued characteristic(s) chosen as ... assessment endpoint(s) ... a quantitative expression of an observed or measured effect of the hazard" (EPA 1989f). The concept of measurement endpoints is not generally applicable to a screening-level assessment because quantitative ecological studies are usually not conducted. Assessment endpoints will be used in this EA indirectly through the use of literature values of chronic and acute toxicity, bioaccumulation, and other effects.

Since it is impossible to evaluate the effects of contamination on all of the potentially exposed ecological receptors occurring at the site, endpoints must be limited to a carefully selected set of potential effects on a few indicator species. Criteria for selection of assessment endpoints for site investigations include the following: social relevance; biological relevance; unambiguous definition; amenability to measurement or prediction; susceptibility to the hazard; and logical relationship to cleanup alternatives (EPA 1989f).

Potential endpoints may be evaluated at the individual, population, community, or ecosystem level. In practical terms, however, good methods and data are available only for the individual, population, or community levels; effects on ecosystems are not normally included as assessment endpoints.

6.10 TOXICITY ASSESSMENT

This subsection of the EA consists of the ecotoxicological profiles of the CPECs selected above. The ecotoxicological profiles briefly summarize the dose-response relationship between exposure magnitudes and effects inducement. Information for the profiles is derived from chronic toxicity tests, which are routinely conducted in laboratories under controlled environmental conditions and with standard test species.

6.10.1 Ecotoxicity Benchmarks

Ecotoxicity benchmarks are calculated from toxicity test dose-response data that describe the relationship between toxicant concentrations and biotic responses. i.e., lethal and/or sublethal effects, such as changes in growth or reproductive rates. The median lethal concentration (LC_{50}) and median effective concentration (EC_{50}) are examples of two commonly used mortality and sublethal effects benchmarks, respectively. The LC_{50} is the toxicant concentration that is lethal for half of the individuals of a test population, whereas the EC_{50} is the toxicant concentration that elicits an adverse response from half of the individuals in the test population.

Benchmarks describing responses to acute toxicity are usually reported as values obtained through direct observations (e.g., LC_{50} s and EC_{50} s), whereas benchmarks describing chronic toxicity are often expressed as estimates of the toxicity threshold concentration (e.g., highest toxicant concentration causing no toxic effects). Chronic toxicity benchmarks are indirect estimates because they are calculated by extrapolating data from relatively short-term (usually from seven to 90 days) toxicity tests to yield long-term relationships.

Lead

Bioaccumulation of lead has been demonstrated for a variety of organisms, with bioconcentration factors typically ranging from 42 to 17,000 (EPA 1986c). Lead has been found in high concentrations in mammals, fish, mollusks-crustacea, plants, mosses, and lichen. This is because lead, and heavy metals in general, bond with amino acids that may be contained in proteins (including enzymes) or polypeptides. This characteristic of heavy metals, in their organic forms, causes them to strongly bind to protein and other biological tissue, increases bioaccumulation, and inhibits excretion. When lead levels reach or exceed the toxicity threshold in the liver and kidneys, excretion and detoxification are affected. In addition to accumulation in soft tissues, toxic levels of lead can accumulate in osseous (bone) tissue.

Lead can be ingested directly from environmental media or indirectly through the food source that has accumulated lead. Heavy metals, including lead, have been shown to accumulate in a variety of wildlife species. In the vicinity of large piles of industrial tailings, water snakes, bullfrogs, muskrats, green-backed herons, and rough-winged and bank swallows

exhibited higher lead levels downgradient of the contamination site. Livers sampled from bullfrogs contained 6.3 ppm wet-weight lead. In four freshwater invertebrate species exposed to lead, bioconcentration factors ranged from 499 to 1,700 (Borgmann, Kramar, and Lovendge 1978; Atchison *et al.* 1977).

Lead also affects several blood parameters in wildlife. Dehydratase levels in blood plasma are inversely related to lead levels. Canvasback ducks with blood lead levels of 200 ppb had reduced dehydratase levels (ALAD enzyme) by 75% (Dieter, Perry, and Mulhern 1976). In canvasback ducks, this reduced level of dehydratase in the blood causes malfunction of the hemoglobin. This, coupled with the fact that ducks have a short erythrocyte half-life of about 40 days, could result in physiological changes in oxygen exchanges from lead exposure.

Water hardness, and the closely related presence of calcium, affect lead toxicity. Daphnids were found to be three times more sensitive to lead in soft water than in hard water (EPA 1980a). Scuds were reported by Spehar *et al.* (1978) to be sensitive to lead. Borgmann, Kramar, and Lovendge (1978) conducted a hard water life cycle study on snails and found that lead levels as low as 19 $\mu\text{g/L}$ significantly decreased survival but not growth or reproduction. Chronic limits, taken from a cumulative percent survival figure, showed no observed effect on survival at 12 $\mu\text{g/L}$ and almost complete mortality at 54 $\mu\text{g/L}$.

Further demonstrating that lead is more chronically toxic in soft water than in hard water is evidence that 60% to 100% of trout exposed to lead levels of 50 $\mu\text{g/L}$ and above, over a 19-month period, developed spinal deformities in hard water. However, when exposed in soft water to lead levels as low as 31 $\mu\text{g/L}$, 44% to 97% of the trout developed spinal deformities. Brook trout have been shown to develop spinal deformities due to lead exposure in a life cycle test (Holcombe *et al.* 1976). Rainbow trout, northern pike, and walleye exposed to lead in early life cycle tests have also shown such deformities.

Calcium seems to decrease lead toxicity by preventing the lead-induced precipitation of mucus in fish. When freshwater fish are exposed to lead levels insufficient to cause death, but sufficient to cause a film of adulterated mucus to form, these organisms develop respiratory problems. This lead-induced film that is particularly prominent over the gills of the fish and the insoluble material interferes with the respiratory function of the gills, causing respiratory distress and potentially death by suffocation. If the film is shed, recovery occurs.

Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) are fairly persistent in the environment due to their chemical properties. PCBs are not likely to leach into the environment due to their extremely low water solubility (solubility of Aroclor 1254 = 0.012 mg/L [EPA 1979]) and their tendency to adsorb to soil organic matter. PCBs bioaccumulate but do not biodegrade or volatilize readily. PCBs can be altered through phototransformation, and, when degradation occurs, it takes place through oxidation, hydrolysis, photodecomposition, or biological metabolism.

There is a direct relationship between the chlorine content of PCBs and their toxicity; PCBs with higher chlorine contents are less easily metabolized. Also, being chlorinated hydrocarbons, PCBs behave similarly to DDT and its metabolites, as well as to other chlorinated insecticides such as Endrin, Dieldrin, and others. Acute toxicity to freshwater aquatic life occurs at concentrations above 2 $\mu\text{g/L}$; however, sensitivity to PCBs varies significantly. For instance, concentrations as low as 0.1 $\mu\text{g/L}$ have killed some species of fish, whereas other species have survived exposure to concentrations above 1,000 ppb (Eisler 1986).

Chronically low levels of PCBs found in sediments will continue to pose a threat to wildlife, although not necessarily a direct lethal threat. PCBs may cause a predisposition to disease as shown by laboratory studies on mallard ducks. Mallards became more susceptible to duck hepatitis virus and suffered higher mortality when exposed to PCBs (Friend and Trainer 1970). Additionally, metabolic rates are affected by PCB contamination and could result in higher energy demand in wildlife species with chronic PCB body burdens. PCBs induce mixed function oxidase activity in the liver; therefore, levels of steroids could be reduced and reproductivity influenced. They also act as synergists for some insecticides in invertebrate systems. When mallard ducks were exposed to DDE, PCBs, and crude oil, egg laying was delayed. Risebrough and Anderson (1975) also reported that when PCBs were added to DDE treatments, there was a decrease in number of intact eggs laid.

Copper

Elevated copper concentrations can affect aquatic ecosystems and cause changes in community structure and nutrient cycling.

their uses are not expected to be negatively affected if the four-day average concentration of zinc ($\mu\text{g/L}$) does not exceed the numerical value of :

$$(0.8473) [\ln(\text{hardness})] + 0.7614 \quad [\text{Eq. 6-7}]$$

more than once every three years on average.

Juveniles and adults are less sensitive to zinc than are earlier stages of saltwater invertebrates and fishes. This sensitivity decreases as salinity increases. It is believed that saltwater aquatic organisms and their uses should not be negatively affected if the four-day average concentration of zinc does not exceed $86 \mu\text{g/L}$ more than once every three years, on average.

Cadmium

Cadmium, a relatively rare heavy metal, is not biologically essential or beneficial. Eisler (1985) reported that cadmium is a known teratogen and carcinogen in fish and wildlife. Freshwater biota are especially sensitive to elevated cadmium concentrations. Cadmium concentrations of $0.8 \mu\text{g/L}$ to $9.9 \mu\text{g/L}$ in water were lethal to several species of aquatic insects, crustaceans, and fish, and concentrations of $0.7 \mu\text{g/L}$ to $5.0 \mu\text{g/L}$ were associated with sublethal effects.

It is conservatively estimated that adverse effects on fish or wildlife are either pronounced or probable when cadmium concentrations exceed $3 \mu\text{g/L}$ in freshwater.

Growth of freshwater aquatic plants is reduced by cadmium at concentrations ranging from $2 \mu\text{g/L}$ to $7,400 \mu\text{g/L}$. These values are in the same range as the acute toxicity values for fish and invertebrate species and are considerably above the chronic values. Bioconcentration factors for cadmium in freshwater range from 164 to 4,190 for invertebrates and from 3 to 2,213 for fishes (EPA 1985a).

Arsenic

As summarized by Eisler (1987a), arsenic is a teratogen and carcinogen to fish and wildlife and can traverse placental barriers and produce fetal death and malformations in many species of mammals. Adverse effects of arsenic on aquatic organisms have been reported at

Message

A few reports are available on the relationship between manganese and environmental media and manganese levels in tissues of aquatic organisms. Mathis (1979) has reported that the manganese content of several species of rooted aquatic plants was proportional to the dissolved manganese concentrations in water, whereas several species of fishes maintained relatively constant manganese levels regardless of concentrations in water. Laboratory exposure of water weeds to manganese at elevated levels indicates that photosynthesis can be inhibited.

DOT and Its Metabolites

0222

chronic toxicity of DDD to sensitive freshwater aquatic life (EPA 1980b; World Health Organization [WHO] 1989).

The sublethal effects of DDT and its metabolites (DDE and DDD) to aquatic organisms include impairment of reproduction and development, cardiovascular modifications, neurological changes, changes in development and behavior, and biochemical alterations.

Information on the sensitivity of aquatic plants species, including algae, to DDT and its metabolites is limited. However, studies with freshwater algae have shown a wide range of sensitivity (EPA 1980b). DDT and its metabolites have been found to reduce photosynthesis. Sublethal effects of DDD in mammals are numerous and include teratogenicity, mutagenicity, and carcinogenicity. In birds, DDD can lower the reproductive rate by causing eggshell thinning and subsequent embryo deaths. Bird species, however, vary greatly in their sensitivity.

Bioaccumulation is an important fate process for DDT and its metabolites in aquatic systems. BCFs range from 10^3 to 10^5 .

Both the acute and chronic toxicities of DDT and its metabolites vary between species of aquatic organisms and life stages with early life stages (e.g., egg and larval) being more sensitive than adults. *Daphnia* reproduction is adversely affected by DDE.

Polynuclear Aromatic Hydrocarbons (PAHs)

Several studies on freshwater invertebrates and lower vertebrates have shown that PAHs can produce cancer-like growths and cause teratogenic and mutagenic effects. Fish exposed to hydrocarbon-contaminated sediments showed a significant number of liver tumors. It appeared that these fish had absorbed PAHs from the sediment (EPA 1982). Data from toxicity bioassays from several studies indicate the general concentration ranges of PAHs that cause toxic effects (acute and chronic).

Reports have shown that PAHs at low concentrations tend to promote growth in bacteria and freshwater algae and can be bioaccumulated in plants, aquatic organisms, and wildlife from intake of contaminated water, sediment, and food. Extensive metabolism of the compounds by higher-trophic level consumers has been demonstrated; therefore, food chain biomagnification of the compounds does not appear to be significant.

Mercury

Once absorbed by organisms, mercury has a variety of deleterious effects on fish and wildlife. These effects have been documented from laboratory studies, but there is a paucity of information on the effects of mercury in field populations. In general, the laboratory data show that mercury is mutagenic, teratogenic, neurotoxic, and carcinogenic, causing adverse effects on cells, organs, reproduction, growth, viability, and behavior. Early developmental stages are generally the most sensitive to the adverse effects of mercury.

A variety of factors affect the toxicity and bioaccumulation of mercury in surface water environments. Toxicity is greatest at high temperatures and in the presence of other metals. Laboratory exposure of fish to mercury in surface water causes sluggishness, loss of appetite, and emaciation. Pathological responses in fish include brain lesions and cataracts (Eisler 1987b).

In freshwater sediments, toxicity to benthic invertebrates occurs at relatively low levels of mercury concentration. For example, studies by OME (1991) indicated that 0.1 mg/kg dry weight is the No Observable Effect Level, 0.12 mg/kg is the Lowest Observable Effect Level, and 2.0 mg/kg is the Limit of Tolerance above which most species are adversely affected. These levels do not take bioaccumulation from sediments into account. Toxicity to benthic invertebrates is likely to vary among sediments.

Nickel

Aquatic organisms have various sensitivities to the acute effects of nickel with the concentrations eliciting the effect ranging from 1,101 $\mu\text{g/L}$ for a cladoceran to 43,420 $\mu\text{g/L}$ for a fish. BCFs for nickel range from 0.8 for fish to 193 for a cladoceran. Toxicity of nickel is correlated with hardness. Data available indicate that toxicity decreases as hardness increases.

Nickel appears to be quite toxic to freshwater algae, with concentrations as low as 50 $\mu\text{g/L}$ producing significant reduction of growth. Patrick, Bott, and Larson (1975) found a decrease in diatom diversity and a shift to green and blue-green algae as a community effect of nickel exposure. Their findings are in agreement with a field study conducted by Spencer and Greene (1981) in which an increase in blue-green algae was observed.

Nickel is highly toxic to single-celled organisms. Bringmann and Kuhn (1959a, 1959b, 1977, and 1981) reported that concentrations of 2.5 to 1500 $\mu\text{g/L}$ resulted in inhibition of algae, bacteria, and protozoans.

Nebeker, Savonen, and Stevens (1985) conducted early life-stage tests with rainbow trout embryos 4 hours and 25 days after fertilization, respectively. In the 4-hour tests, weight was significantly reduced by all tested concentrations including the lowest of 35 $\mu\text{g/L}$. In the 25-day tests, survival was reduced at nickel concentrations of 134 $\mu\text{g/L}$ and higher.

Chromium

A temporary reduction in growth occurred in brook trout and rainbow trout treated with concentrations of chromium as low as 264 $\mu\text{g/L}$. The bioconcentration factor obtained with the trout is less than 3. Also, daphnids gave chronic values that range from <2.5 to 40 $\mu\text{g/L}$.

The toxicity of chromium increases as pH is lowered or as hardness is lowered or both. Although there are exceptions, softer surface waters usually have a lower pH than harder surface waters.

The effect concentrations of chromium range from 2 $\mu\text{g/L}$ for blue-green alga to 7,800 $\mu\text{g/L}$ for a diatom. Toxicity of chromium to diatoms was tested at two hardnesses (Academy of Natural Sciences 1960). The geometric mean of the concentrations causing a 50% reduction in growth was 245 $\mu\text{g/L}$ at the lower water hardness and 335 $\mu\text{g/L}$ at the higher hardness.

Toxicity of chromium to bluegill was determined by the Academy of Natural Sciences (1960) at two hardnesses. The 96-hour LC50s were 113,000 $\mu\text{g/L}$ in soft water and 135,000 $\mu\text{g/L}$ in hard water.

Hogendoorn-Roozmond *et al.* (1978) reported the toxicity of chromium at two pH levels. They reported that young rainbow trout were more sensitive to chromium at pH 6.9 than pH 7.9. Van der Putte *et al.* (1982) found that chromium is more toxic at pH 6.5 than at pH 7.8.

- If the concentration is equal to or above the LOT, then significant impairment of sediments used by benthos is expected.

6.11 RISK CHARACTERIZATION

In this section, the ecological risks posed by CPECs at the Conrail Site are identified and summarized. In Section 6.11.1, risks are estimated using hazard index (HI) ratios calculated using estimated exposure and toxicity benchmark values for each receptor. The ratios are summarized and the principal uncertainties of the assessment are discussed in Section 6.11.2. The ecological significance of the findings is discussed in Section 6.11.3.

6.11.1 Hazard Index (HI) Ratio

The risks of site contamination were quantified by calculating an HI ratio for each CPEC and pathway. The HIs were calculated as follows:

$$HI = ED/TRV \quad [Eq. 6-8]$$

where: HI = Hazard index;
 ED = Estimated dosage or concentrations in medium; and
 TRV = Toxicity reference value.

TRVs are the benchmark criteria described in Section 6.10.2 of this EA. The estimated dosage (ED) or concentrations were either the maximum or the mean concentration of a chemical in the location of concern. The HIs for aquatic organisms in surface water are provided in Table 6-34. The HIs for sediment dwelling (benthic) organisms are provided in Table 6-35.

An HI greater than unity (1) is typically selected as the point of departure for the potential for chronic toxicological risk to a given ecological receptor.

6.11.2 Summary of Risks and Uncertainties

For benthic organisms in the sediment and surface water, exposure to CPECs, especially cadmium, copper, lead, zinc, fluorene, and 4-methylphenol, may result in adverse effects. The HIs for the maximum concentration of these CPECs are as follows: cadmium-16.00, copper-24.16, lead-39.68, zinc-56.91, fluorene-10.24, and 4-methylphenol-8.50.

Arsenic, chromium, manganese, mercury, nickel, DDT, DDD, and Aroclor 1254 also had HIs greater than unity (see Table 6-2), but the magnitudes of the HIs were lower than the HIs for the previously stated CPECs. The maximum concentrations were all located in the westernmost pond. As a result, resident biota receiving chronic exposures to the sediment and surface water may be at risk for toxic effects by the current contaminant levels of all CPECs.

Uncertainty arises from applying generic nationally based benchmarks as TRVs for a specific area. The benchmarks were designed to be very conservative to protect sensitive species of pristine environments; however, the area of concern in this EA is an urban area surrounded by agricultural land, and is contaminated by various sources and therefore probably lacks sensitive biota.

A related source of error originates from the notion that benthic organisms tend to be more resistant to chemicals than are column biota (EPA 1989f). AWQC are derived using both benthic and column biota, so use of AWQC to derive sediment criteria inherently causes an overestimation of risk. This source of uncertainty errs in the conservative (ecosystem-protective) direction and does not compromise the assessment.

6.11.3 Conclusions

Potential risks to the benthic organisms from the CPECs present in the ponds are assessed to be moderate to high. CPECs were detected only in the westernmost pond that contained the outfall from the site. Currently, there are no species of special concern residing in the ponds, or in the area, that would be affected by the potential risks posed by CPECs in the ponds. Although there is a potential for some adverse effects from exposure to CPECs to individuals residing in the ponds, it is unclear what effects these affected individuals may have on the overall population structure, habitat diversity, or ecosystem as a whole. It should be kept in mind that these habitats are also affected by the land uses on site and in the areas surrounding the site (i.e., agricultural, residential, and industrial). Physical disturbances from these land uses may have (or have had in the past) more influence on populations in the ponds than the site-related chemical stressors.

Table 6-1
ADEQUACY OF DETECTION LIMITS FOR CHEMICALS
DETECTED AT THE CONRAIL SITE

Chemical	Soils/Sediments			Groundwater/Surface Water		
	Sample Detection Limits (µg/kg)	Cancer Risk ^a	Hazard Index	Sample Detection Limits (µg/L)	Cancer Risk ^b	Hazard Index
VOCs						
Acetone	10 - 4,500	--	4.89 E-09	10 - 50	--	2.74 E-03
Benzene	5 - 1,300	1.10 E-12	--	10 - 50	3.39 E-06 ^c	--
Bromodichloromethane	5 - 1,300	4.54 E-12	1.23 E-07	10 - 50	1.52 E-05 ^c	1.37 E-02
Bromomethane	10 - 1,700	--	--	10 - 50	--	--
Bromoform	5 - 1,300	2.76 E-13	1.23 E-08	10 - 50	9.24 E-07	1.37 E-02
2-Butanone	10 - 4,500	--	9.78 E-09	10 - 50	--	5.48 E-03
Carbon disulfide	5 - 1,300	--	--	10 - 50	--	2.74 E-03
Carbon tetrachloride	5 - 1,300	4.54 E-12	3.50 E-07	10 - 100	1.52 E-05 ^c	3.91 E-01
Chlorobenzene	5 - 1,300	--	1.23 E-08	10 - 50	--	1.37 E-02
Chloroethane	10 - 1,700	--	1.63 E-09	10 - 50	--	9.13 E-05
Chloroform	5 - 1,300	2.13 E-13	2.45 E-07	10 - 50	7.14 E-07	2.74 E-02
Dibromochloromethane	5 - 1,300	2.93 E-12	1.23 E-08	10 - 50	9.83 E-06 ^c	1.37 E-02
1,1-Dichloroethane	5 - 1,300	--	2.45 E-09	10 - 50	--	2.74 E-03
1,2-Dichloroethane	5 - 1,300	3.18 E-12	--	10 - 50	1.06 E-05 ^c	--

Table 6-1
ADEQUACY OF DETECTION LIMITS FOR CHEMICALS
DETECTED AT THE CONRAIL SITE

Chemical	Soils/Sediments			Groundwater/Surface Water		
	Sample Detection Limits (µg/kg)	Cancer Risk ^a	Hazard Index	Sample Detection Limits (µg/L)	Cancer Risk ^b	Hazard Index
1,1-Dichloroethene	5 - 1,300	2.09 E-11	2.72 E-07	10 - 50	7.02 E-05 ^c	3.04 E-02
1,2-Dichloroethene (total)	5 - 1,300	--	2.45 E-08	10 - 50	--	2.74 E-02
1,2-Dichloropropene	5 - 1,300	2.37 E-12	--	10 - 50	7.96 E-06 ^c	--
1,3-Dichloropropene	5 - 1,300	6.28 E-12	8.17 E-07	10 - 50	2.11 E-05 ^c	9.13 E-01
Ethylbenzene	5 - 1,300	--	2.45 E-08	10 - 50	--	2.74 E-03
2-Hexanone (MBK)	10 - 1,700	--	--	10 - 50	--	--
4-Methyl-2-pentanone	10 - 1,700	--	9.78 E-09	10 - 50	--	5.48 E-03
Methyl chloride	10 - 1,700	9.09 E-13	--	10 - 50	1.52 E-06 ^c	--
Methylene chloride	5 - 1,300	2.6 E-13	4.08 E-08	10 - 50	8.77 E-07	4.57 E-03
Styrene	5 - 1,300	1.05 E-12	1.23 E-09	10 - 50	3.31 E-06 ^c	1.37 E-03
1,1,2,2-Tetrachloroethane	5 - 1,300	6.98 E-12	--	10 - 50	2.34 E-05 ^c	--
Tetrachloroethene	5 - 1,300	1.78 E-12	2.45 E-08	10 - 50	5.97 E-06 ^c	2.74 E-02
Toluene	5 - 1,300	--	1.23 E-09	10 - 50	--	1.37 E-03
1,1,1-Trichloroethane	5 - 1,350	--	2.72 E-09	10 - 50	--	3.04 E-03
1,1,2-Trichloroethane	5 - 1,300	1.99 E-12	6.13 E-08	10 - 50	6.67 E-06 ^c	6.85 E-02

6-85

0230

Table 6-1 ADEQUACY OF DETECTION LIMITS FOR CHEMICALS DETECTED AT THE CONRAIL SITE						
Chemical	Soils/Sediments			Groundwater/Surface Water		
	Sample Detection Limits (µg/kg)	Cancer Risk ^a	Hazard Index	Sample Detection Limits (µg/L)	Cancer Risk ^b	Hazard Index
Trichloroethene	5 - 1,300	3.84 E-13	--	10 - 1,000	1.29 E-06 ^c	--
Vinyl acetate	10 - 1,300	--	--	10 - 50	--	--
Vinyl chloride	10 - 1,700	1.33 E-10	--	10 - 50	2.22 E-04 ^c	--
Xylenes	5 - 1,300	--	6.13 E-10	10 - 50	--	1.37 E-04

^a Risk-based concentration derived by calculating the risk of the lowest detection limit using the soil ingestion pathway.

^b Risk-based concentration derived by calculating the risk of the lowest detection limit using the drinking water ingestion pathway.

^c Cancer risk value exceeds EPA's 10⁻⁶ limit.

Key:

-- Not applicable.

Source: Ecology and Environment, Inc. 1994.

DATA QUALIFIERS AND THEIR EFFECT ON DATA USE IN THE RISK ASSESSMENT

0232

Table 6-2 DATA QUALIFIERS AND THEIR EFFECT ON DATA USE IN THE RISK ASSESSMENT				
Qualifier	Definition	Uncertain Identity?	Uncertain Concentration?	Include Data in Quantitative Risk Assessment?
+	Correlation coefficient for MSA was less than 0.995	No	Yes	Yes

Source: Ecology and Environment, Inc. 1994.

**SUMMARY OF SOIL BORING DATA (0.5 TO 12 FEET BGS) FOR
ORGANIC CHEMICALS (µg/kg)**

^a Chemical of potential concern.
^b Detected infrequently and at low concentration.
^c Concentration is the average of two concentrations from a boring and its duplicate.

Table 6-4

SUMMARY OF GROUNDWATER DATA FOR
ORGANIC CHEMICALS (µg/L)

Chemical	Frequency		Range Detected			
	On-Site Samples	Background Samples	Minimum Concentration Detected	Sample Location	Maximum Concentration Detected	Sample Location
Volatile Organic Compounds						
Acetone ^a	10/64	0/13	2	MW42-1	29.5	MW41-2
Benzene ^{a,b}	2/64	0/13	2	MW23D-3	3	MW25-1
Bromodichloromethane ^{a,b}	1/64	0/13	0.9	MW7-1	0.9	MW7-1
Bromomethane ^b	1/64	0/13	5.5	MW46S-1	5.5	MW46S-1
2-Butanone (MEK) ^{a,b,c}	3/64	0/13	4	MW15-3	7	MW21S-3
Carbon disulfide ^b	3/64	0/13	2	MW41-1	3	MW3-1/ MW46S-1
Carbon tetrachloride ^a	26/64	0/13	2	MW32I-2	94,500	MW46S-1
Chloroform ^a	18/64	0/13	1	MW5D-1	525	MW46S-1
1,1-Dichloroethane ^a	1/64	0/13	3	MW30BR-1	3	MW30BR-1
1,1-Dichloroethene ^a	2/64	0/13	7	MW9-1	56	MW41-1
1,2-Dichloroethene ^a	14/64	0/13	2	MW23S-3	230	MW41-1
Methyl chloride ^a	3/64	0/13	6	MW14-1	74	MW46S-1

Table 6-4
SUMMARY OF GROUNDWATER DATA FOR
ORGANIC CHEMICALS ($\mu\text{g/L}$)

Chemical	Frequency		Range Detected			
	On-Site Samples	Background Samples	Minimum Concentration Detected	Sample Location	Maximum Concentration Detected	Sample Location
Methylene chloride ^a	3/64	0/13	3.5	MW41-2	5	MW32S-1A
Tetrachloroethene (PCE) ^a	8/64	0/13	3	MW32S-2	10	MW32S-1A
Toluene ^{b,c}	1/64	0/13	3	MW331-1A	3	MW331-1A
1,1,1-Trichloroethane ^a	6/64	0/13	3	MW23D-2	76	MW9-2
1,1,2-Trichloroethane ^b	1/64	0/13	2	MW41-1	2	MW41-1
Trichloroethene (TCE) ^a	43/64	0/13	1	MW48-1	15,000	MW41-2
Vinyl chloride ^a	1/64	0/13	6.5	MW51-1	6.5	MW51-1

^a Chemical of potential concern.

^b Detected infrequently and at low concentration.

^c Common laboratory artifact.

Source: Ecology and Environment, Inc. 1994.

Table 6-5					
SUMMARY OF SURFACE WATER AND SEDIMENT DATA FOR ORGANIC CHEMICALS					
Chemical	Frequency of Detection	Range Detected			
		Minimum Concentration Detected	Sample Location	Maximum Concentration Detected	Sample Location
Surface Water Volatile Organic Compounds (µg/L)					
Trichloroethene (TCE) ^b	1/25	35	SW15	35	SW15
Sediment Volatile Organic Compounds (µg/kg)					
Acetone ^c	5/25	17	SD06	100	SD08
2-Butanone (MEK) ^b	1/25	44	SD14	44	SD14
Carbon tetrachloride ^b	1/25	67	SD10	67	SD10
1,2-Dichloroethene ^b	1/25	32	SD15	32	SD15
Methylene chloride ^c	11/25	6	SD27	46	SD08
Toluene ^c	5/25	2	SD13	15	SD05
Trichloroethene (TCE) ^b	2/25	12	SD15	100	SD10

^b Detected infrequently and at low concentration.

^c Common laboratory artifact.

Source: Ecology and Environment, Inc. 1994.

Table 6-6 SUMMARY OF CHEMICALS OF POTENTIAL CONCERN FOR THE CONRAIL SITE		
Chemical	Soil	Groundwater
VOCs		
Acetone	X	X
Benzene	X	X
Bromodichloromethane		X
2-Butanone (MEK)	X	X
Carbon tetrachloride	X	X
Chloroform	X	X
1,1-Dichloroethane		X
1,1-Dichloroethane		X
1,2-Dichloroethane	X	X
Ethylbenzene	X	
4-Methyl-2-pentanone (MIBK)	X	
Methyl chloride		X
Methylene chloride	X	X
Tetrachloroethane (PCE)	X	X
Toluene	X	
1,1,1-Trichloroethane	X	X
Trichloroethane	X	X
Vinyl chloride	X	X
Xylenes	X	

Source: Ecology and Environment, Inc. 1994.

Table 6-7

**SUMMARY OF POTENTIALLY COMPLETE EXPOSURE PATHWAYS
 EVALUATED FOR THE CONRAIL SITE**

Receptor	Exposure Point	Exposure Pathway	Exposure Media	Derivation of Exposure-Point Concentrations	Basis for Source Concentration
Facility Workers and Visitors	Classification Yard of the Conrail facility	Vapor inhalation	Airborne Vapors	Modeling of vapor emissions and air dispersion	Measured concentrations in sub-surface soil (0.5-12 feet) and groundwater in the classification yard
		Dermal contact	Subsurface Soil	Subsurface soil concentrations	Measured concentrations in sub-surface soil (0.5-12 feet) in the classification yard
		Incidental ingestion			
Nearby Residents	Plume 1 Areas (County Road 1, Charles Avenue, and Vista Avenue areas)	Drinking water ingestion	Groundwater	Groundwater concentrations	Measured groundwater concentrations in plume flowing north and west from the Conrail facility towards St. Joseph River
		Dermal contact while showering			
		Indoor air vapor inhalation	Indoor air	Modeling of soil gas intrusion into residential basements	Measured groundwater concentrations in plume flowing north and west from the Conrail facility towards St. Joseph River
		Vapor inhalation while showering	Vapors	Shower stall volatilization model	Measured groundwater concentrations in plume flowing north and west from the Conrail facility towards St. Joseph River

Table 6-7

**SUMMARY OF POTENTIALLY COMPLETE EXPOSURE PATHWAYS
EVALUATED FOR THE CONRAIL SITE**

Receptor	Exposure Point	Exposure Pathway	Exposure Media	Derivation of Exposure-Point Concentrations	Basis for Source Concentration
Nearby Residents (Cont.)	Plume 2 Area (LaRue Street area)	Drinking water ingestion	Groundwater	Groundwater concentrations	Measured groundwater concentrations in plume flowing north from the east portion of the Conrail facility towards St. Joseph River
		Dermal contact while showering	Vapors	Shower stall volatilization model	Measured groundwater concentrations in plume flowing north from the east portion of the Conrail facility towards St. Joseph River
		Vapor inhalation while showering			
		Indoor air vapor inhalation	Indoor air	Modeling of soil gas infiltration into residential buildings	Measured groundwater concentrations in plume flowing north from the east portion of the Conrail facility towards St. Joseph River

Source: Ecology and Environment, Inc. 1994.

6-95

0240

<p>Table 6-8</p> <p>MAXIMUM AND UCL CONCENTRATIONS OF</p> <p>SUBSURFACE SOILS (0.5-12 FEET BGS)</p>		
Chemical	Maximum Concentration (mg/kg)	Upper Confidence Limit (mg/kg)
Acetone	5,400	5,943
Benzene	19	--
2-Butanone	2,500	--
Carbon tetrachloride	29	--
Chloroform	3	--
1,2-Dichloroethene	5,600	--
Ethylbenzene	4,700	--
4-methyl-2-pentanone	6,700	--
Methylene chloride	71.5	25.3
Tetrachloroethene	7	6.69
Toluene	145	25.5
Trichloroethene	14,000	7,707
Vinyl chloride	8	--
Xylenes	12,000	--

-- Not calculated due to limited number of samples.

Source: Ecology and Environment, Inc. 1994.

0241

Table 6-9

**FACILITY WORKER EXPOSURE:
PATHWAY 1A - INHALATION OF AIRBORNE VAPORS**

Equation:

$$\text{Intake (mg/kg-day)} = \frac{\text{CA} \times \text{IR} \times \text{ET} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

where:

- CA = Contaminant concentration in air (mg/m³)
- IR = Inhalation rate (m³/hour)
- ET = Exposure time (hours/day)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CA	Adult	CT/RME	Modeled value based on UCL or maximum observed soil and groundwater concentrations
IR	Adult	CT/RME	20 m ³ /workday (EPA 1991a)
ET	Adult	CT/RME	8 hours/day (EPA 1991a)
EF	Adult	CT/RME	250 days/year (EPA 1991a)
ED	Adult	CT	5 years (USDOL 1987)
		RME	25 years (95th percentile amount of time working at a single location (EPA 1991a))
BW	Adult	CT/RME	70 kg (adult, average (EPA 1991a))
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

- CT = Central Tendency.
- RME = Reasonable maximum exposure.
- UCL = Upper 95 percent confidence limit on the arithmetic average.
- USDOL = United States Department of Labor.

Source: Ecology and Environment, Inc. 1994.

Table 6-10

**RESIDENTIAL WATER USAGE:
PATHWAY 2A - INGESTION OF CHEMICALS IN DRINKING WATER
(AND BEVERAGES MADE USING DRINKING WATER)
ADULTS**

Equation:

$$\text{Intake (mg/kg-day)} = \frac{\text{CW} \times \text{IR} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

where:

- CW = Chemical concentration in water (mg/L)
- IR = Ingestion rate (L/day)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CW	Adult	CT/RME	Maximum concentrations detected in monitoring wells
IR	Adult	CT	1.4 L/day (EPA 1993b)
		RME	2.0 L/day (90th percentile [EPA 1991a])
EF	Adult	CT/RME	350 days/year (EPA 1991a)
ED	Adult	CT	9 years (50th percentile time at one residence [EPA 1993b])
		RME	30 years (90th percentile time at one residence [EPA 1991a])
BW	Adult	CT/RME	70 kg (average [EPA 1991a])
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

- CT = Central Tendency.
- RME = Reasonable maximum exposure.
- Source: Ecology and Environment, Inc. 1994.

Table 6-11

**RESIDENTIAL WATER USAGE:
PATHWAY 2B - DERMAL CONTACT WITH CHEMICALS DURING SHOWERING
ADULTS**

Equation:

$$\text{Absorbed dose (mg/kg-day)} = \frac{\text{CW} \times \text{PC} \times \text{SA} \times \text{ET} \times \text{EF} \times \text{ED} \times \text{CF}}{\text{BW} \times \text{AT}}$$

where:

- CW = Chemical concentration in water (mg/L)
- PC = Chemical-specific dermal permeability constant (cm/hr)
- SA = Skin surface area available for contact (cm²)
- ET = Exposure time (hours/day)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- CF = Volumetric conversion factor for water (1 L/1,000 cm³)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Reference/Source)
CW	Adult	CT/RME	Maximum concentrations in groundwater
PC	Adult	CT/RME	Chemical-specific values used (EPA 1992c)
SA	Adult	CT	20,000 cm ² (average total body [EPA 1992c])
		RME	23,000 cm ² (95th percentile total body [EPA 1992c])
ET	Adult	CT/RME	0.2 hour/day (12 minutes; 90th percentile [EPA 1989d])
EF	Adult	CT/RME	350 days/year (EPA 1991a)
ED	Adult	CT	9 years (50th percentile time at one residence [EPA 1993b])
		RME	30 years (90th percentile time at one residence [EPA 1991a])
BW	Adult	CT/RME	70 kg (average [EPA 1991a])

Table 6-11			
RESIDENTIAL WATER USAGE: PATHWAY 2B - DERMAL CONTACT WITH CHEMICALS DURING SHOWERING ADULTS			
Variable	Receptor	Case	Value (Rationale/Source)
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

CT = Central Tendency.
 RME = Reasonable maximum exposure.

Source: Ecology and Environment, Inc. 1994.

Table 6-12

**RESIDENTIAL WATER USAGE:
PATHWAY 2C - INHALATION OF AIRBORNE (VAPOR-PHASE) CHEMICALS
DURING SHOWERING
ADULTS**

Equation:

$$\text{Intake (mg/kg-day)} = \frac{\text{CA} \times \text{IR} \times \text{ET} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

where:

- CA = Contaminant concentration in air (mg/m³)
- IR = Inhalation rate (m³/hour)
- ET = Exposure time (hours/day)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CA	Adult	CT/RME	Value modeled from maximum concentration in groundwater
IR	Adult	CT/RME	0.83 m ³ /hr (all age groups (EPA 1991a))
ET	Adult	CT/RME	0.2 hr/day (12 minutes; 90th percentile (EPA 1989d))
EF	Adult	CT/RME	350 days/year (EPA 1991c)
ED	Adult	CT	9 years (50th percentile time at one residence (EPA 1993b))
		RME	30 years (90th percentile time at one residence (EPA 1991a))
BW	Adult	CT/RME	70 kg (average (EPA 1991a))
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

- CT = Central Tendency.
- RME = Reasonable maximum exposure.
- Source: Ecology and Environment, Inc. 1994.

**RESIDENTIAL EXPOSURE:
PATHWAY 2D - INHALATION OF INDOOR AIRBORNE VAPORS
ADULTS**

where:

CA = Contaminant concentration in air (mg/m^3)
IR = Inhalation rate (m^3/hour)
ET = Exposure time (hours/day)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CA	Adult	CT/RME	Value modeled from maximum concentration in groundwater
IR	Adult	CT/RME	0.63 m ³ /hr (EPA 1991a)
ET	Adult	CT/RME	24 hours/day (EPA 1991a)
EF	Adult	CT/RME	350 days/year (EPA 1991a)
ED	Adult	CT	9 years (50th percentile time at one residence [EPA 1993b])
		RME	30 years (90th percentile time at one residence [EPA 1991a])
BW	Adult	CT/RME	70 kg (average [EPA 1991a])
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

CT = Central Tendency.
RME = Reasonable maximum exposure.
Source: Ecology and Environment, Inc. 1994.

**FUTURE SCENARIO FACILITY WORKER EXPOSURE:
PATHWAY 3A - INGESTION OF CHEMICALS IN SUBSURFACE SOIL**

WAGE

- CS = Chemical concentration in soil (mg/kg)
IR = Ingestion rate (mg soil/day)
CF = Conversion factor (10^{-6} kg/mg)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CS	Adult	CT/RME	Modeled value based on UCL or maximum observed soil concentrations
IR	Adult	CT	30 mg/day (EPA 1991a)
		RME	480 mg/day (EPA 1993b)
EF	Adult	CT	65 days/year
		RME	250 days/year (EPA 1991a)
ED	Adult	CT/RME	1 year
BW	Adult	CT/RME	70 kg (average [EPA 1991a])
AT	Adult	CT/RME	Pathway-specific period for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

CT = Central Tendency.
RME = Reasonable maximum exposure.
UCL = Upper 95 percent confidence limit on the arithmetic average.

Source: Ecology and Environment, Inc. 1994.

Table 6-15

**FUTURE SCENARIO FACILITY WORKER EXPOSURE:
PATHWAY 3B - DERMAL CONTACT WITH CHEMICALS IN SUBSURFACE SOIL**

Equation: Absorbed Dose (mg/kg-day) =
$$\frac{CS \times ABS \times CF \times SA \times AF \times EF \times ED}{BW \times AT}$$

where:

- CS = Chemical concentration in soil (mg/kg)
- ABS = Fraction absorbed (unitless)
- CF = Conversion factor (10^{-6} kg/mg)
- SA = Skin surface area available for contact (cm^2/event)
- AF = Soil to skin adherence factor (mg/cm^2)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CS	Adult	CT/RME	Modeled value based on UCL or maximum observed soil concentrations
ABS	Adult	CT/RME	Chemical-specific value
SA	Adult	CT	5,000 cm^2 (EPA 1992c)
		RME	5,800 cm^2 (EPA 1992c)
AF	Adult	CT/RME	1.0 mg/cm^2 (EPA 1992c)
EF	Adult	CT	65 days/year
		RME	250 days/year (EPA 1991a)
ED	Adult	CT/RME	1 year
BW	Adult	CT/RME	70 kg (EPA 1991a)
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

- CT = Central Tendency.
- RME = Reasonable maximum exposure.
- UCL = Upper 95 percent confidence limit on the arithmetic average.

Source: Ecology and Environment, Inc. 1994.

Table 6-16

**FUTURE SCENARIO FACILITY WORKER EXPOSURE:
PATHWAY 3C - INHALATION OF VAPORS FROM
NEWLY EXPOSED CONTAMINATED SUBSURFACE SOILS**

Equation:

$$\text{Intake (mg/kg-day)} = \frac{\text{CA} \times \text{IR} \times \text{ET} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

where:

- CA = Contaminant concentration in air (mg/m³)
- IR = Inhalation rate (m³/hour)
- ET = Exposure time (hours/day)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged, in days)

Variable	Receptor	Case	Value (Rationale/Source)
CA	Adult	CT	Chemical-specific concentration modeled from UCL or maximum soil concentration averaged over 65 days
		RME	Chemical specific concentration modeled from UCL or maximum soil concentration averaged over 250 days
IR	Adult	CT/RME	20 m ³ /workday (EPA 1991a)
ET	Adult	CT/RME	8 hours/day (EPA 1991a)
EF	Adult	CT	65 days/year
		RME	250 days/year (EPA 1991a)
ED	Adult	CT/RME	1 year
BW	Adult	CT/RME	70 kg (EPA 1991a)
AT	Adult	CT/RME	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70-year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

Key:

- CT = Central Tendency.
- RME = Reasonable maximum exposure.
- UCL = Upper 95 percent confidence limit on the arithmetic average.

Source: Ecology and Environment, Inc. 1994.

Table 6-17

**DERMAL PERMEABILITY CONSTANTS
FOR WATER AND DERMAL ABSORPTION
COEFFICIENTS FOR SOIL USED
IN THE CONRAIL SITE RISK ASSESSMENT**

Chemical	Water Dermal Permeability Constants	Reference	Soil Dermal Absorption Coefficients	Reference
Acetone	0.00057	EPA 1992c	0.001	EPA 1993
Benzene	0.11000	EPA 1992c	0.001	EPA 1993
Bromodichloromethane	0.00580	EPA 1992c	—	—
2-Butanone	0.00110	EPA 1992c	0.001	EPA 1993
Carbon Tetrachloride	0.02200	EPA 1992c	0.001	E & E 1993
Chloroform	0.00890	EPA 1992c	0.001	E & E 1993
1,1-Dichloroethane	0.00890	EPA 1992c	—	—
1,1-Dichloroethene	0.01600	EPA 1992c	—	—
1,2-Dichloroethene	0.01000	EPA 1992c	0.001	EPA 1993
Ethylbenzene	—	—	0.030	EPA 1993
Methyl Chloride	0.00420	EPA 1992c	—	—
Methylene Chloride	0.00450	EPA 1992c	0.001	E & E 1993
Methyl Isobutyl Ketone	—	—	0.030	E & E 1993
Tetrachloroethene	0.04800	EPA 1992c	0.030	EPA 1993
Toluene	0.04500	EPA 1992c	0.030	EPA 1993
1,1,1-Trichloroethane	0.01700	EPA 1992c	0.001	EPA 1993
Trichloroethene	0.01600	EPA 1992c	0.001	EPA 1993
Vinyl Chloride	0.00730	EPA 1992c	0.001	EPA 1993
Xylenes (total)	—	—	0.030	EPA 1993

Key:

— Not applicable.

EPA WEIGHT-OF-EVIDENCE CATEGORIES FOR CHEMICAL CARCINOGENICITY

Group	Description
A	Human Carcinogen - sufficient evidence from epidemiological studies to support a causal association between exposure and cancer.
B	Probable Human Carcinogen.
B1	At least limited evidence of carcinogenicity to humans from epidemiological studies.
B2	Usually a combination of sufficient evidence of carcinogenicity in animals and inadequate evidence of carcinogenicity in humans.
C	Possible Human Carcinogen - limited evidence of carcinogenicity in animals in the absence of human data.
D	Not Classified - inadequate evidence of carcinogenicity in animals.
E	No Evidence of Carcinogenicity for Humans - no evidence of carcinogenicity in at least two adequate animal tests in different species or in both epidemiological and animal studies.

Source: EPA, 1986, "Guidelines for Carcinogen Risk Assessment," Federal Register 51:33992-34012.

Table 6-19
TOXICITY VALUES FOR POTENTIAL CARCINOGENIC EFFECTS: CONFIDENCE IN CLASSIFICATION AND IN SLOPE FACTORS (SFs)

Chemical	Route	Slope Factor (SF) (mg/kg-day)	Weight-of-Evidence Classification	Type of Cancer	SF Basis/SF Source
Acetone	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS
Benzene	Oral Inhalation	2.9 E-02 2.9 E-02	A A	Leukemia Leukemia	IRIS IRIS
Bromodichloromethane	Oral Inhalation	6.2 E-02 6.3 E-03	B2 B2	Kidney NA	Gavage/IRIS NA/IRIS
2-Butanone	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS
Carbon tetrachloride	Oral Inhalation	1.30 E-01 5.25 E-02	B2 B2	Liver Liver	Gavage/IRIS Gavage/IRIS
Chloroform	Oral Inhalation	6.10 E-03 8.10 E-02	B2 B2	Liver Liver	Gavage/IRIS Gavage/IRIS
1,1-Dichloroethane	Oral Inhalation	ND ND	C C	NA NA	NA/IRIS NA/IRIS
1,1-Dichloroethene	Oral Inhalation	6.00 E-01 1.80 E-01	C C	Adrenal Kidney	Drinking water/IRIS Inhalation/IRIS
1,2-Dichloroethene	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS
Ethylbenzene	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS

Table 6-19

TOXICITY VALUES FOR POTENTIAL CARCINOGENIC EFFECTS: CONFIDENCE IN CLASSIFICATION AND IN SLOPE FACTORS (SFs)

Chemical	Route	Slope Factor (SF) (mg/kg-day)	Weight-of-Evidence Classification	Type of Cancer	SF Basis/SF Source
Methyl Chloride	Oral Inhalation	1.30 E-02 6.30 E-03	C C	Kidney Kidney	Drinking water/IRIS Inhalation/IRIS
Methylene chloride	Oral Inhalation	7.50 E-03 1.60 E-03	B2 B2	Liver Adenomas	Drinking water/IRIS Inhalation/IRIS
4-Methyl-2-pentanone	Oral Inhalation	ND ND	-- --	NA NA	NA/IRIS and HEAST NA/IRIS and HEAST
Tetrachloroethene	Oral Inhalation	5.2 E-02 2.0 E-03	B2 B2	Liver Liver	HEAST HEAST
Toluene	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS
1,1,1-Trichloroethane	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS
Trichloroethane	Oral Inhalation	1.10 E-02 6.00 E-03	B2 B2	Liver Lung	HEAST HEAST
Vinyl chloride	Oral Inhalation	1.90 E+00 3.0 E-01	A A	Lung Liver	Diet/HEAST Inhalation/HEAST
Xylene	Oral Inhalation	ND ND	D D	NA NA	NA/IRIS NA/IRIS

6-109

0254

Table 6-19 (Cont.)

Key:

- HEAST = Health Effects Assessment Summary Tables
- IRIS = Integrated Risk Information System
- NA = Not applicable
- ND = Not determined

Source: Ecology and Environment, Inc. 1994.

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6-110

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0255

UNCERTAINTY FACTORS (SAFETY FACTORS) USED IN THE DERIVATION OF REFERENCE DOSES

Uncertainty Factor	Condition of Use
10	A 10-fold uncertainty factor is used with valid experimental results on appropriate durations of exposures of humans.
100	A 100-fold uncertainty factor is used when human data are not available and extrapolation is made from valid results of long-term animal studies.
1,000	A 1,000-fold uncertainty factor is used when human data are not available and extrapolation is made from animal studies of less than chronic exposure.
1-10	An additional uncertainty factor from 1 to 10 when using a lowest observed adverse effect level (LOAEL) instead of a no observed adverse effect level (NOAEL).
Intermediate uncertainty factor	Other uncertainty factors used, according to scientific judgment, when justified.

Source: EPA, 1986, "Guidelines for Carcinogen Risk Assessment," *Federal Register* 51:33992-34012.

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
Acetone	Oral	Chronic	1.00 E-01	Low	Increased liver and kidney weight - nephrotoxicity	Gavage/IRIS	UF = 1,000 MF = 1
		Subchronic	1.00 E+00	NS	Increased liver and kidney weight - nephrotoxicity	Gavage/HEAST	UF = 100 MF = 1
	Inhalation	Chronic	1.00 E-01	NS	NA	Extrapolated from oral value	-- --
		Subchronic	1.00 E+00	NS	NA	Extrapolated from oral value	-- --
Benzene	Oral	Chronic	ND	NS	NA	NA	NA
		Subchronic	ND	NS	NA	NA	NA
	Inhalation	Chronic	ND	NS	NA	NA	NA
		Subchronic	ND	NS	NA	NA	NA
Bromodichloromethane	Oral	Chronic	1.4 E-03	Medium	Hyperplasia	Gavage/IRIS	UF = 1,000 MF = 1
		Subchronic	1.4 E-03	High	Gas, respiratory	Inhalation/IRIS	UF = 100 MF = 1

Key at end of table.

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6-112

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0257

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
Bromodichloromethane (Cont.) 2-Butanone (Methyl Ethyl Ketone)	Inhalation	Chronic	2.00 E-02	NS	Cytomegaly	Gavage/HEAST	UF = 1,000 MF = 1
		Subchronic	2.00 E-02	NS	NA	Extrapolated from oral subchronic	..
	Oral	Chronic	6.00 E-01	Low	Decreased birth weight	Feeding Study/IRIS	UF = 3,000 MF = 1
		Subchronic	2.00 E-01	NS	No Effect	Inhalation/HEAST	UF = 1,000 MF = 1
	Inhalation	Chronic	3.00 E-01	NS	No effect	Animal study/IRIS	UF = 1,000 MF = 1
		Subchronic	3.00 E+00	NS	No Effect	Inhalation/HEAST	UF = 3,000 MF = 1
Carbon Tetrachloride	Oral	Chronic	7.00 E-04	NS	Liver Lesions	Oral/HEAST	UF = 1,000 MF = 1
		Subchronic	7.00 E-03	NS	Liver Lesions	Oral/HEAST	UF = 1,000 MF = 1

Key at end of table.

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
Carbon Tetrachloride (Cont.)	Inhalation	Chronic	7.00 E-04	NS	NA	Extrapolated from oral	--
		Subchronic	7.00 E-03	NS	NA	Extrapolated from oral	--
Chloroform	Oral	Chronic	1.00 E-02	Medium	Cyst formation on liver	Oral/IRIS	UF = 1,000 MF = 1
		Subchronic	1.00 E-02	NS	Liver Lesions	Oral/HEAST	UF = 1,000 MF = 1
	Inhalation	Chronic	1.00 E-02	NS	NA	Extrapolated from oral	--
		Subchronic	1.00 E-02	NS	NA	Extrapolated from oral	--
1,1-Dichloroethane	Oral	Chronic	1.00 E-01	NS	No effect	Oral/HEAST	UF = 1,000 MF = 1
		Subchronic	1.00 E+00	NS	No effect	Oral/HEAST	UF = 100 MF = 1

Key at end of table.

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6-114

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0259

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
1,1-Dichloroethene (Cont.)	Inhalation	Chronic	1.40 E-01	NS	None observed	Inhalation/HEAST	UF = 1,000 MF = 1
		Subchronic	1.40 E+00	NS	None observed	Inhalation/HEAST	UF = 100 MF = 1
1,1-Dichloroethene	Oral	Chronic	9.00 E-03	Medium	Hepatic lesions	Oral Bioassay/IRIS	UF = 1,000 MF = 1
		Subchronic	9.00 E-03	NS	Hepatic lesions	Drinking water/HEAST	UF = 1,000 MF = 1
	Inhalation	Chronic	9.00 E-03	NS	NA	Extrapolated from oral	-- --
		Subchronic	9.00 E-03	NS	NA	Extrapolated from oral	-- --
1,2-Dichloroethene	Oral	Chronic	1.00 E-02	NS	Decreased hemocrit	Gavage/HEAST	UF = 3,000 MF = 1
		Subchronic	1.00 E-01	NS	Decreased hemoglobin	Gavage/HEAST	UF = 300 MF = 1

Key at end of table.

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
1,2-Dichloroethene (Cont.)	Inhalation	Chronic	1.00 E-02	NS	NA	Extrapolated from oral	--
		Subchronic	1.00 E-01	NS	NA	Extrapolated from oral	--
Ethylbenzene	Oral	Chronic	1.00 E-01	Low	Liver and kidney toxicity	Oral/IRIS	UF = 1,000 MF = 1
		Subchronic	1.00 E-01	NS	Hepatotoxicity, nephrotoxicity	Oral/HEAST	UF = 100 MF = 1
	Inhalation	Chronic	2.90 E-01	Low	Developmental toxicity	Air/IRIS	UF = 300 MF = 1
		Subchronic	2.90 E-01	NS	Developmental toxicity	Air/HEAST	UF = 300 MF = 1
Methyl Chloride	Oral	Chronic	ND	NS	NA	NA	--
		Subchronic	ND	NS	NA	NA	--

Key at end of table.

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6-116

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0261

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
Methyl Chloride (Cont.)	Inhalation	Chronic	ND	NS	NA	NA	--
		Subchronic	ND	NS	NA	NA	--
Methylene Chloride	Oral	Chronic	6.00 E-02	Medium	Liver toxicity	Drinking water/IRIS	UF = 100 MF = 1
		Subchronic	6.00 E-02	NS	Liver toxicity	Drinking water/HEAST	UF = 100 MF = 1
	Inhalation	Chronic	8.57 E-01	NS	NA	Inhalation/HEAST	UF = 100 MF = 1
		Subchronic	8.57 E-01	NS	NA	Inhalation/HEAST	UF = 100 MF = 1
4-Methyl-2-pentanone	Oral	Chronic	5.00 E-02	NS	Liver and kidney effects	Gavage/HEAST	UF = 1,000 MF = 1
		Subchronic	5.00 E-01	NS	Liver and kidney effects	Gavage/HEAST	UF = 100 MF = 1

Key at end of table.

Table 6-21							
TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS							
Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
4-Methyl-2-pentanone (Cont.)	Inhalation	Chronic	2.30 E-02	NS	Increased liver and kidney weight	Inhalation/HEAST	UF = 1,000 MF = 1
		Subchronic	2.30 E-01	NS	Increased liver and kidney weight	Inhalation/HEAST	UF = 100 MF = 1
Tetrachloroethene	Oral	Chronic	1.00 E-02	Medium	Hepatotoxicity	Gavage/IRIS	UF = 1,000 MF = 1
		Subchronic	1.00 E-01	NS	Hepatotoxicity	Oral/HEAST	UF = 100 MF = 1
	Inhalation	Chronic	1.00 E-02	NS	NA	Extrapolated from oral	-- --
		Subchronic	1.00 E-01	NS	NA	Extrapolated from oral	-- --
Toluene	Oral	Chronic	2.00 E-01	Medium	Changes in liver and kidney weight	Gavage/IRIS	UF = 1,000 MF = 1
		Subchronic	2.00 E-00	Medium	Changes in liver and kidney weight	Gavage/HEAST	UF = 100 MF = 1

Key at end of table.

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6-118

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0263

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
Toluene (Cont.)	Inhalation	Chronic	1.20 E-01	Medium	Neurological (CNS effects), eye and nose irritation	Inhalation/TRIS	UF = 300 MF = 1
		Subchronic	6.00 E-01	Medium	Neurological (CNS effects), eye and nose irritation	Inhalation/HEAST	UF = 100 MF = 1
1,1,1-Trichloroethane	Oral	Chronic	9.00 E-02	Medium	Hepatotoxicity	Oral/TRIS	UF = 1,000 MF = 1
		Subchronic	9.00 E-01	NS	Hepatotoxicity	Oral/HEAST	UF = 100 MF = 1
	Inhalation	Chronic	3.00 E-01	NS	Hepatotoxicity	Inhalation/HEAST	UF = 1,000 MF = 1
		Subchronic	3.00 E+00	NS	Hepatotoxicity	Inhalation/HEAST	UF = 100 MF = 1
Trichloroethene	Oral	Chronic	ND	NS	NA	NA	--
		Subchronic	ND	NS	NA	NA	--

Key at end of table.

Table 6-21							
TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS							
Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-clay)				
Trichloroethene (Cont.)	Inhalation	Chronic	ND	NS	NA	NA	--
		Subchronic	ND	NS	NA	NA	--
Vinyl Chloride	Oral	Chronic	ND	NS	NA	NA	--
		Subchronic	ND	NS	NA	NA	--
	Inhalation	Chronic	ND	NS	NA	NA	--
		Subchronic	ND	NS	NA	NA	--
Xylenes	Oral	Chronic	2.00 E+00	Medium	CNS effects	Gavage/IRIS	UF = 100 MF = 1
		Subchronic	4.00 E+00	Medium	CNS effects	Gavage/HEAST	UF = 100 MF = 1

Key at end of table.

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6-120

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0265

Table 6-21

TOXICITY VALUES FOR POTENTIAL NONCARCINOGENIC EFFECTS

Reference Dose (RfD)				Confidence Level	Critical Effect	RfD Basis/Source	Uncertainty Factor (UF) and Modifying Factor (MF)
Chemical	Route	Type	Value (mg/kg-day)				
Xylenes (Cont.)	Inhalation	Chronic	2.00 E+00	NS	NA	Extrapolated from oral	--
		Subchronic	4.00 E+00	NS	NA	Extrapolated from oral	--

Key:

- CNS = Central Nervous System
- HEAST = Health Effects Assessment Summary Tables
- IRIS = Integrated Risk Information System
- NA = Not Applicable
- ND = Not Determined
- NS = Not Specified
- = Not Applicable

6-121

0266

Table 6-22					
SUMMARY OF ESTIMATED EXCESS CANCER RISKS ASSOCIATED WITH THE CONRAIL SITE					
Pathway	Location	Case	Receptor	Risk Contributions by Exposure Route ^a	Significant Risk Contributions by Chemical ^a
			Adult		
CURRENT CONDITIONS					
Worker Exposure					
Inhalation of airborne contaminants	Conrail Facility	CT RME	6.80 E-04 3.40 E-03	Vapor inhalation - 100%	Carbon tetrachloride - 97% Vinyl chloride - 2.5%
Nearby Residential Exposure					
Groundwater usage	Plume 1	CT RME	2.15 E-03 9.00 E-03	Ingestion - 67% Dermal Contact in Shower - 3% Vapor Inhalation - 31%	Ingestion: Carbon tetrachloride - 42% 1,1-DCE - 4% Trichloroethene - 19% Vinyl chloride - 2% Dermal: Carbon tetrachloride - 2% Inhalation: Carbon tetrachloride - 17% Chloroform - 2% 1,1-DCE - 1% Trichloroethene - 11%
Inhalation of indoor airborne contaminants from groundwater vapors	Plume 1	CT RME	5.11 E-05 1.71 E-04	Inhalation - 100%	Carbon tetrachloride - 93% 1,1-DCE - 5% Trichloroethene - 2%

Table 6-22
SUMMARY OF ESTIMATED EXCESS CANCER RISKS
ASSOCIATED WITH THE CONRAIL SITE

Pathway	Location	Case	Receptor	Risk Contributions by Exposure Route ^a	Significant Risk Contributions by Chemical ^a
			Adult		
Groundwater usage	Plume 2	CT RME	4.23 E-04 1.45 E-03	Ingestion - 8% Dermal Contact - 1% Vapor inhalation - 90%	Ingestion: Carbon tetrachloride - 8% Dermal: Carbon tetrachloride - 1% Inhalation: Carbon tetrachloride - 1% Chloroform - 1% Benzene - 90%
Inhalation of indoor airborne contaminants from groundwater vapors	Plume 2	CT RME	3.31 E-06 1.11 E-05	Inhalation - 100%	Carbon tetrachloride - 95% Chloroform - 2% Trichloroethene - 3%
POTENTIAL FUTURE CONDITIONS					
Worker Exposure					
Soil contact	Conrail Facility	CT RME	1.91 E-05 8.50 E-05	Dermal Contact - 100%	Trichloroethene - 81% Vinyl chloride - 14%
Inhalation of airborne contaminants	Conrail Facility	CT RME	4.64 E-07 8.83 E-07	Inhalation - 100%	None

^a = RME case for receptor showing greatest risk. These columns are independent of each other; both refer to total receptor risks.
CT = Central Tendency.
RME = Reasonable Maximum Exposure.
- = Not applicable.
Source: Ecology and Environment, Inc. 1994.

6-123

0268

Table 6-23

**SUMMARY OF ESTIMATED HAZARD INDICES FOR NONCARCINOGENIC EFFECTS
ASSOCIATED WITH THE CONRAIL SITE**

Pathway	Location	Case	Receptor	Risk Contributions by Exposure Route ^a	Significant Risk Contributions by Chemical ^a
			Adult		
CURRENT CONDITIONS					
Worker Exposure					
Inhalation of airborne contaminants	Conrail Facility	CT RME	25 253	Inhalation - 100%	Carbon tetrachloride - 100%
Nearby Residential Exposure					
Groundwater usage	Plume 1	CT RME	172 201	Ingestion - 49% Dermal Contact - 2% Vapor Inhalation - 49%	Ingestion: Carbon tetrachloride - 48% Dermal: Carbon tetrachloride - 2% Inhalation: Carbon tetrachloride - 48%
Inhalation of indoor airborne contaminants from groundwater vapors	Plume 1	CT RME	10.1 10.1	Inhalation - 100%	Carbon tetrachloride - 100%
Groundwater usage	Plume 2	CT RME	3.22 4.16	Ingestion - 72% Dermal Contact - 4% Vapor inhalation - 24%	Ingestion: Carbon tetrachloride - 72% Dermal Contact: Carbon tetrachloride - 4% Inhalation: Acetone - 15% Carbon tetrachloride - 7%

Table 6-23

**SUMMARY OF ESTIMATED HAZARD INDICES FOR NONCARCINOGENIC EFFECTS
ASSOCIATED WITH THE CONRAIL SITE**

Pathway	Location	Case	Receptor	Risk Contributions by Exposure Route ^a	Significant Risk Contributions by Chemical ^a
			Adult		
Inhalation of indoor airborne contaminants from groundwater vapors	Plume 2	CT RME	6.70 E-01 6.70 E-01	Inhalation - 100%	None
POTENTIAL FUTURE RECEPTORS					
Worker Exposure					
Soil contact	Conrail Facility	CT RME	1.08 E+01 4.81 E+01	Dermal Contact - 100%	Acetone - 6% Carbon tetrachloride - 5% 1,2-DCB - 67%
Inhalation of airborne contaminants	Conrail Facility	CT RME	3.82 E-02 1.91 E-02	Inhalation - 100%	Ethylbenzene - 6% 4-Methyl-2-pentanone - 16%

- ^a = RME case for receptor showing greatest risk. These columns are independent of each other; both refer to total receptor risks.
CT = Central Tendency.
RME = Reasonable Maximum Exposure.
- = Not applicable.

Source: Ecology and Environment, Inc. 1994.

6-125

0270

SPECIAL SPECIES OF CONCERN LOCATED WITHIN 2 MILES OF THE CONRAIL SITE

Common Name	Scientific Name	Location	Year Last Observed	Status
Michaux's Stichwort	<i>Arenaria stricta</i>	37N4E Sec 2	1945	State Rare
Cooper's Hawk	<i>Accipiter cooperii</i>	37N4E Sec 4 SWQ 37N4E Sec 5 NWQ 37N4E Sec 13 38N4E Sec EH SWQ	1986 1985 1986 1986	State Special Concern
Grooved Yellow Flax	<i>Linum sulcatum</i>	37N4E Sec 9 NEQ	1935	State Rare
Badger	<i>Taxidea taxus</i>	38N4E Sec 28 & 32	1989	State Threatened

0271

Table 6-25

RESULTS OF IDNR FISH SAMPLING EFFORT

Common Name	Scientific Name	Bittersweet Bridge Area	Cabot Creek Location Area
Golden Redhorse	<i>Moxostoma erythrurum</i>	X	X
Largemouth Bass	<i>Micropterus salmoides</i>	X	X
Bluegill	<i>Lepomis macrochirus</i>	X	X
Carp	<i>Cyprinus carpio</i>	X	X
Smallmouth Bass	<i>Micropterus dolomieu</i>	X	X
Northern Hog Sucker	<i>Hypentelium nigricans</i>	-	X
Pumpkinseed Sunfish	<i>Lepomis gibbosus</i>	X	X
White Sucker	<i>Catostomus commersoni</i>	X	X
Longnose Gar	<i>Lepisosteus osseus</i>	X	X
Bowfin	<i>Amia calva</i>	-	X
Longear Sunfish	<i>Lepomis longiears</i>	-	X
Spotted Sucker	<i>Moxostoma valenciennesi</i>	X	X
Green Sunfish	<i>Lepomis cyanellus</i>	X	-
Yellow Perch	<i>Perca flavescens</i>	-	X
Golden Shiner	<i>Notemigonus crysoleucas</i>	-	X
Logperch	<i>Percina caprodes</i>	X	X
Common Shiner	<i>Notropis cornutus</i>	-	X
Shorthead Redhorse	<i>Moxostoma macrolepidotum</i>	X	X
Black Crappie	<i>Pomoxis nigromaculatus</i>	X	X
White Crappie	<i>Pomoxis annularis</i>	X	X
Channel Catfish	<i>Ictalurus punctatus</i>	X	X
Rock Bass	<i>Ambloplites rupestris</i>	X	-
Quillback	<i>Cariacus cyprinus</i>	X	-
Yellow Bullhead	<i>Ictalurus nebulosus</i>	X	-
Brown Bullhead	<i>Ictalurus nebulosus</i>	X	X
Spotfin Shiner	<i>Notropis spilopterus</i>	-	X

Table 6-25			
RESULTS OF IDNR FISH SAMPLING EFFORT			
Common Name	Scientific Name	Bittersweet Bridge Area	Cobus Creek Location Area
Spottail Shiner	<i>Notropis hudsonius</i>	-	X
Northern Pike	<i>Esox niger</i>	X	X

Key:

- X = Present at sample location.
- = Not present at sample location.

Source: Compiled by Ecology and Environment, Inc. 1994.

Table 6-26
SUMMARY OF WETLANDS SURROUNDING CONRAIL SITE

Wetland Symbol	Wetland Habitat Description	Acres North of Facility*	Acres South of Facility*
PFO1A	Palustrine forested. Broad-leaved Deciduous. Temporarily Flooded	1.5, 5, 1.5	3
POWGx	Palustrine Open Water. Intermittently Exposed. Excavated	1.5, 1	1
PEMFx	Palustrine. Emergent. Semipermanently Flooded. Excavated	1, 1	—
PABFx	Palustrine. Aquatic Bed. Semipermanently Flooded. Excavated	1	—
PEMC	Palustrine. Emergent. Seasonally Flooded	1	1
P SS1/EM B	Palustrine. Mixed Scrub-Shrub Broad Leaved Deciduous and Emergent. Saturated	12	—
PFO1B	Palustrine Forested. Broad-Leaved Deciduous. Saturated	12	—
POWG	Palustrine Open Water. Intermittently Exposed	—	1, 1.2
PAB4G	Palustrine. Aquatic Bed. Floating Vascular. Intermittently Exposed	—	0.5
P FO/SS 1C	Palustrine. Mixed Forested and Scrub-Shrub. Broad-Leaved Deciduous. Seasonally Flooded	—	10, 1
P FO/SS 1A	Palustrine. Mixed Forested and Scrub-Shrub. Broad-Leaved Deciduous. Temporarily Flooded	—	13, 1.5

* Presented are the approximate acreage of individual wetlands in the noted areas.
— Not present.

Table 6-27

SCREENING OF SURFACE WATER INORGANIC RESULTS

Inorganic	Chronic AWQC (µg/L)	Ponds		Baugo Bay Background		Baugo Bay		St. Joseph River Background		St. Joseph River Downstream of Crawford Ditch	
		Result Range (µg/L)	Number Exceeding Criterion	Result Range (µg/L)	Number Exceeding Criterion	Result Range (µg/L)	Number Exceeding Criterion	Result Range (µg/L)	Number Exceeding Criterion	Result Range (µg/L)	Number Exceeding Criterion
Lead	3.2	ND - 41.1	1/3	ND - 1.1	0/3	ND - 2.8	0/5	ND - 1.1	0/6	ND - 5.2	1/8
Zinc	110	ND - 155	1/3	ND - 13.8	0/3	12.9 - 50.8	0/5	ND	0/6	ND - 15.2	0/8

ND = Not detected.

Source: Ecology and Environment, Inc. 1994.

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6-130

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0275

Table 6-28 PUBLISHED INTERIM SEDIMENT QUALITY CRITERIA		
Contaminant	SQC ^a (µg/kg)	Reference
Pyrene	13,100	EPA 1988
Benzo(a)anthracene	13,200	EPA 1988
Fluoranthene	18,000	EPA 1988
Benzo(a)pyrene	10,630	EPA 1988
Acenaphthene	7,330	EPA 1988
Dieldrin	199	EPA 1988
Endrin	10.4	EPA 1988
Phenanthrene	1,390	EPA 1988
α-Chlordane	62	OMGE 1991
γ-Chlordane	66	OMGE 1991
4,4'-DDT	8.28	EPA 1988
4,4'-DDD	90	OMGE 1991
4,4'-DDE	213	OMGE 1991
Aroclor 1254	196	EPA 1988
Aldrin	1,284	OMGE 1991

^a Values in this column are based on a 1 % Total Organic Carbon (TOC).

Source: Compiled by Ecology and Environment, Inc. 1994.

Table 6-29

CALCULATED SEDIMENT QUALITY CRITERIA

Chemical	FCV ($\mu\text{g/L}$)		Log K_{ow}		K_{oc}^a (L/kg)	SQC ^b ($\mu\text{g/kg}$)
	Value	Source	Value	Source		
4-Methylphenol	40	AQ	1.94	HE	80.78	32.31
2-Methylphenol	200	AQ	1.95	HE	82.63	165.26
Fluorene	0.156	AQ	4.20	HE	1.3455 E+4	20.99
Benzo(b)fluoranthene	1.40	AQ	6.06	HE	9.06275 E+5	12,688
Benzo(k)fluoranthene	1.40	AQ	6.06	HE	9.06275 E+5	12,688
Chrysene	0.7	AQ	5.61	HE	3.27273 E+5	2,291
Indeno(1,2,3-cd)pyrene	0.156	AQ	6.58	HE	2.940492 E+6	4,587
Benzo(g,h,i)perylene	0.156	AQ	6.51	HE	2.509632 E+6	3,915
Naphthalene	620	AWQC	3.30	HE	1.755 E+3	10,879
Anthracene	12.7	AQ	4.45	HE	2.3693 E+4	3,009

^a $\text{Log } K_{oc} = 0.00028 + -0.983(\text{Log } K_{ow})$

^b $\text{SQC} = f_{oc} \cdot K_{oc} \cdot \text{FCV}$; based on 1% TOC

Key:

- AQ = AQUIRE Data base
- AWQC = Chronic Ambient Water Quality Criteria for Protection of Aquatic Life
- HE = HEAST Data base

Source: Compiled by Ecology and Environment, Inc. 1994.

Table 6-30

SCREENING OF SEDIMENT SAMPLES^a

Contaminant	SQC (µg/kg)	Ponds		Bauge Bay Background		Bauge Bay		St. Joseph River Background		St. Joseph River Downstream of Crawford Ditch	
		Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria
Pyrene	13,100	ND-6,000	0/3	ND	0/3	ND-910	0/5	ND-1,200	0/6	ND-1,400	0/8
Benzo(a)anthracene	13,200	ND-1,935	0/3	ND	0/3	ND-450	0/5	ND-690	0/6	ND-650	0/8
Fluoranthene	18,800	ND-1,165	0/3	ND	0/3	ND-650	0/5	ND-1400	0/6	ND-980	0/8
Benzo(a)pyrene	10,630	ND-1,505	0/3	ND	0/3	ND	0/5	ND-720	0/6	ND-430	0/8
Acenaphthalene	7,330	ND	0/3	ND	0/3	ND	0/5	ND-160	0/6	ND	0/8
Dieldrin	199	ND	0/3	ND	0/3	ND-4.1	0/5	ND	0/6	ND	0/8
Endrin	10.4	ND-4.5	0/3	ND	0/3	ND	0/5	ND	0/6	ND	0/8
Phenanthrene	1,390	ND	0/3	ND	0/3	ND-350	0/5	ND-1,300	0/6	ND-460	0/8
γ-Chlordane	62	ND-7.3	0/3	ND	0/3	ND-1.6	0/5	ND	0/6	ND-0.61	0/8
β-Chlordane	66	ND-12.4	0/3	ND	0/3	ND-3	0/5	ND-1.2	0/6	ND-2.4	0/8
1,4'-DDT	8.28	ND-35.5	1/3	ND	0/3	ND	0/5	ND	0/6	ND-2.8	0/8
1,4'-DDD	90	ND-230	1/3	ND	0/3	ND	0/5	ND	0/6	ND-1.9	0/8
1,4'-DDE	213	ND-27.5	0/3	ND	0/3	ND	0/5	ND	0/6	ND-6.5	0/8
Arochlor 1254	196	ND-1,500	1/3	ND	0/3	ND-130	0/5	ND-47	0/6	ND-190	0/8
Aldrin	1,284	ND-4.2	0/3	ND	0/3	ND	0/5	ND	0/6	ND	0/8
4-Methylphenol	32.31	ND-340	1/3	ND	0/3	ND	0/5	ND-120	1/6	ND-76	1/8

Table 6-30

SCREENING OF SEDIMENT SAMPLES^a

Contaminant	SQC (µg/kg)	Ponds		Baugo Bay Background		Baugo Bay		St. Joseph River Background		St. Joseph River Downstream of Crawford Ditch	
		Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria
2-Methylphenol	165.26	ND	0/3	ND	0/3	ND	0/3	ND	0/6	ND-78	0/8
Fluorene	20.99	ND-215	1/3	ND	0/3	ND	0/3	ND-160	1/6	ND	0/8
Benzo(b)fluoranthene	12,688	ND-2,750	0/3	ND	0/3	ND-770	0/3	ND-730	0/6	ND-1,100	0/8
Benzo(k)fluoranthene	12,688	ND-1,320	0/3	ND	0/3	ND-390	0/3	ND-460	0/6	ND-370	0/8
Chrysene	2,291	ND-2,130	0/3	ND	0/3	ND-550	0/3	ND-630	0/6	ND-710	0/8
Indeno(1,2,3)pyrene	4,587	ND-1,875	0/3	ND	0/3	ND	0/3	ND-300	0/6	ND-130	0/8
Benzo(g,h,i)perylene	3,915	ND-1,682	0/3	ND	0/3	ND	0/3	ND-420	0/6	ND	0/8
Naphthalene	10,879	ND-265	0/3	ND	0/3	ND	0/3	ND	0/6	ND	0/8
Anthracene	3,009	ND-341	0/3	ND	0/3	ND	0/3	ND-220	0/6	ND	0/8
Arsenic	5,500	1,100- 8,650	1/3	460-1,600	0/3	370-14,900	3/3	2,800- 10,500	4/6	2,300- 20,500	0/8
Cadmium	1,000	ND- 16,000	1/3	ND	0/3	ND	0/3	ND	0/6	ND-4,500	2/8
Chromium	31,000	4,300- 77,350	1/3	4,600- 6,600	0/3	2,800- 34,900	1/3	6,100- 33,500	1/6	7,500- 38,900	3/8
Copper	25,000	5,700- 604,000	1/3	2,400- 3,700	0/3	2,500- 51,400	3/3	3,300- 44,300	1/6	2,700- 60,700	3/8

Table 6-30

SCREENING OF SEDIMENT SAMPLES^a

Contaminant	SQC (µg/kg)	Ponds		Bauge Bay Background		Bauge Bay		St. Joseph River Background		St. Joseph River Downstream of Crawford Ditch	
		Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria	Result Range (µg/kg)	Number Exceeding Criteria
Lead	31,000	3,600- 1,230,000	1/3	1,500- 14,000	0/3	1,700- 35,700	1/5	5,400- 57,900	1/6	1,700- 84,000	3/8
Manganese	457,000	148,000- 788,000	1/3	102,000- 109,000	0/3	71,200- 1,860,000	3/5	81,000- 1,470,000	2/6	93,900- 2,330,000	3/8
Mercury	120	ND-370	1/3	ND	0/3	ND-300	1/5	ND-2,800	1/6	ND-270	1/8
Nickel	31,000	6,200- 150,850	1/3	4,200- 7,900	0/3	4,300- 30,200	0/5	5,500- 22,700	0/6	5,000- 26,500	0/8
Zinc	110,000	21,600- 6,260,000	1/3	10,400- 18,300	0/3	10,100- 194,000	3/5	14,300- 154,000	2/6	11,300- 198,000	3/8

^a Results of duplicates were averaged for final result.

Key:

ND = Not detected

Source: Ecology and Environment, Inc. 1994.

OME PROVINCIAL SEDIMENT QUALITY GUIDELINES

Contaminant	No Observed Effect Level	Lowest Observed Effect Level	Limit of Tolerance
Arsenic	4.0	5.5	33.0
Cadmium	0.6	1.0	10.0
Chromium	22.0	31.0	111
Copper	15.0	25.0	114
Iron (%)	2.0	3.0	4.0
Lead	23.0	31.0	250
Manganese	400	457	1110
Mercury	0.1	0.12	2.0
Nickel	15.0	31.0	90.0
Zinc	65.0	110	800

Source: Compiled by Ecology and Environment, Inc. 1994.

Table 6-32

**T-TEST OF THE DIFFERENCE BETWEEN BACKGROUND SAMPLE AND
DOWNSTREAM SAMPLE INORGANIC ANALYTICAL RESULTS FOR BAUGO BAY**

Contaminant	Baugo Bay Background			Baugo Bay			F-test ^a	t-Value	Significant Difference ^b
	Sample Size	Mean (mg/kg)	SD _{n-1}	Sample Size	Mean (mg/kg)	SD _{n-1}			
Arsenic	3	1.02	0.57	5	5.25	5.68	9.96	1.23 ^c	No
Chromium	3	5.57	1.00	5	15.26	12.17	12.17	1.32 ^c	No
Copper	3	3.00	0.65	5	19.30	19.58	30.13	2.04 ^d	No
Lead	3	5.80	7.10	5	15.96	13.82	1.95	1.15 ^c	No
Manganese	3	108.33	6.03	5	626.24	705.51	117.00	1.64 ^d	No
Mercury	3	0.100	0	5	0.14	0.09	0	-	No
Nickel	3	5.70	1.95	5	15.34	10.80	5.55	1.48 ^d	No
Zinc	3	13.67	4.12	5	77.28	74.61	18.11	1.43 ^d	No

- ^a $F(df_4, 2) = 19.25$
^b 95% Confidence Limit.
^c Equation 6-4 was used.
^d Equation 6-5 was used.
 - Not Applicable.

Source: Ecology and Environment, Inc. 1994.

Table 6-33									
T-TEST OF THE DIFFERENCE BETWEEN BACKGROUND SAMPLE AND DOWNSTREAM SAMPLE INORGANIC ANALYTICAL RESULTS FOR THE ST. JOSEPH RIVER									
Contaminant	St. Joseph Background			St. Joseph River			F-Test ^a	t-Value ^b	Significant Difference
	Sample Size	Mean (mg/kg)	SD _{n-1}	Sample Size	Mean (mg/kg)	SD _{n-1}			
Arsenic	6	6.78	3.25	8	8.05	7.24	2.22	0.40	No
Chromium	6	14.33	10.49	8	18.83	13.27	1.27	0.68	No
Copper	6	13.95	15.92	8	22.17	24.15	1.52	0.72	No
Lead	6	19.88	20.82	8	28.17	30.37	1.08	0.57	No
Manganese	6	585	563.88	8	622.98	788.74	1.40	0.10	No
Zinc	6	57.27	54.33	8	74.68	77.23	1.42	0.47	No

^a $F(df_{7,5}) = 10.46$.

^b Equation 6-4 was used.

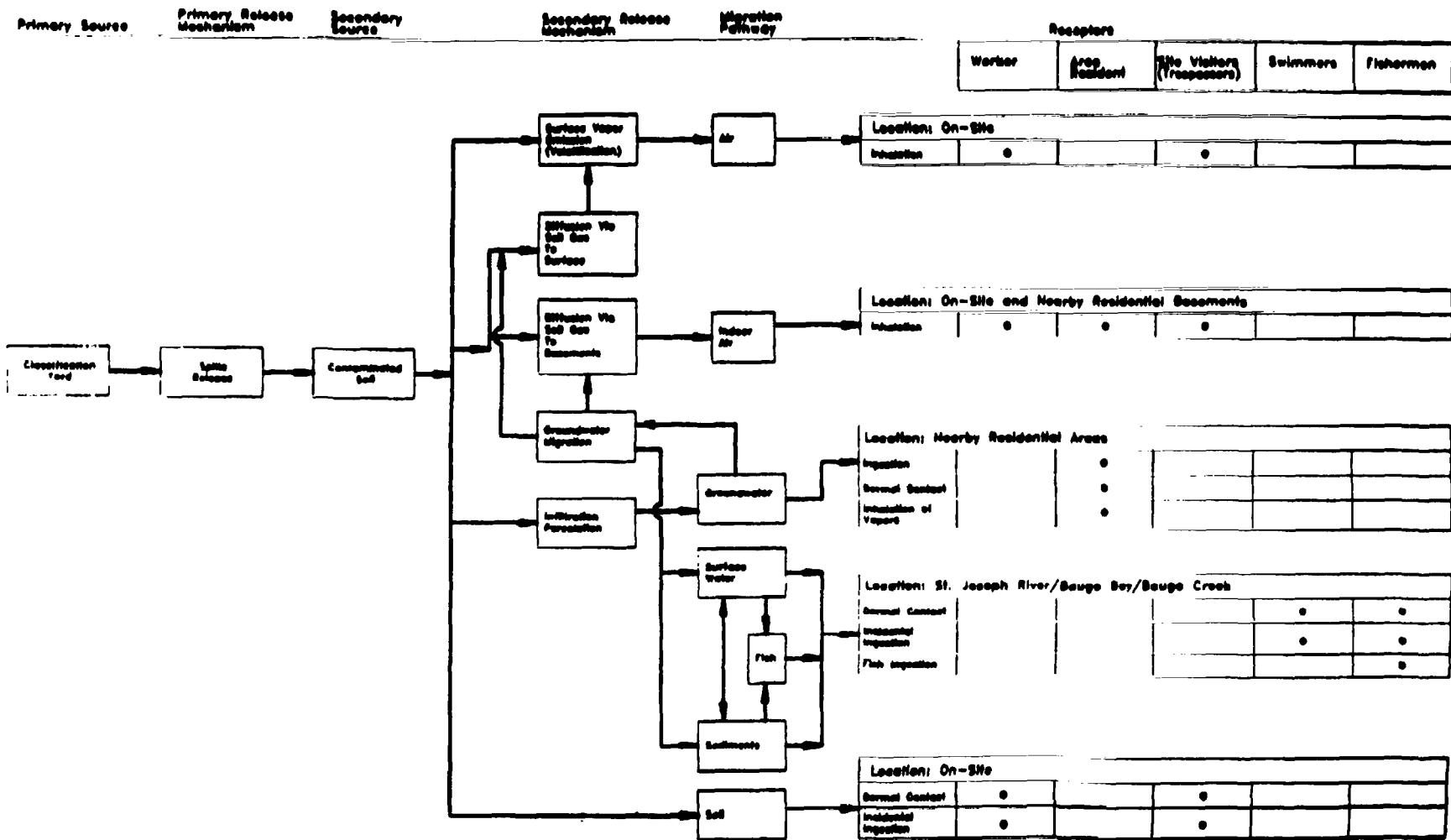
Source: Ecology and Environment, Inc. 1994.

Table 6-34					
SUMMARY OF HAZARD INDICES FOR AQUATIC BIODIVERSITY IN SURFACE WATER					
Contaminant	Benchmark Criteria ($\mu\text{g/L}$)	Maximum Concentration ($\mu\text{g/L}$)	Hazard Index	Mean Concentration ($\mu\text{g/L}$)	Hazard Index
Lead	3.2	25	7.81	9.33	2.92
Zinc	110	105.95	0.96	45.55	0.41

Source: Ecology and Environment, Inc. 1994.

Table 6-35					
SUMMARY OF HAZARD INDICES FOR AQUATIC BIOTA IN SEDIMENT					
Contaminant	Benchmark Criteria (µg/kg)	Maximum Concentration (µg/kg)	Hazard Index	Mean Concentration (µg/kg)	Hazard Index
Arsenic	5,500	8,650	1.57	2,750	0.50
Cadmium	1,000	16,000	16.00	5,667	5.67
Chromium	31,000	77,350	2.49	30,733	0.99
Copper	25,000	604,000	24.16	262,633	10.50
Lead	31,000	1,230,000	39.68	412,833	13.32
Manganese	457,000	788,500	1.73	384,000	0.84
Mercury	120	370	3.08	190	1.58
Nickel	31,000	150,850	4.87	55,917	1.80
Zinc	110,000	6,260,000	56.91	2,101,966	19.11
DDT	8.28	35.5	4.29	13.3	1.61
DDD	90	230	1.08	78.13	0.87
4-Methylphenol	32.31	340	8.50	223	6.91
Fluorene	20.99	215	10.24	182	8.69
Aroclor 1254	196	1,500	7.65	515	2.63

Source: Ecology and Environment, Inc. 1994.



ecology and environment

FIGURE 6-1

CONCEPTUAL SITE
MODEL: HUMAN
HEALTH EVALUATION
GENERAL SITE
6-141

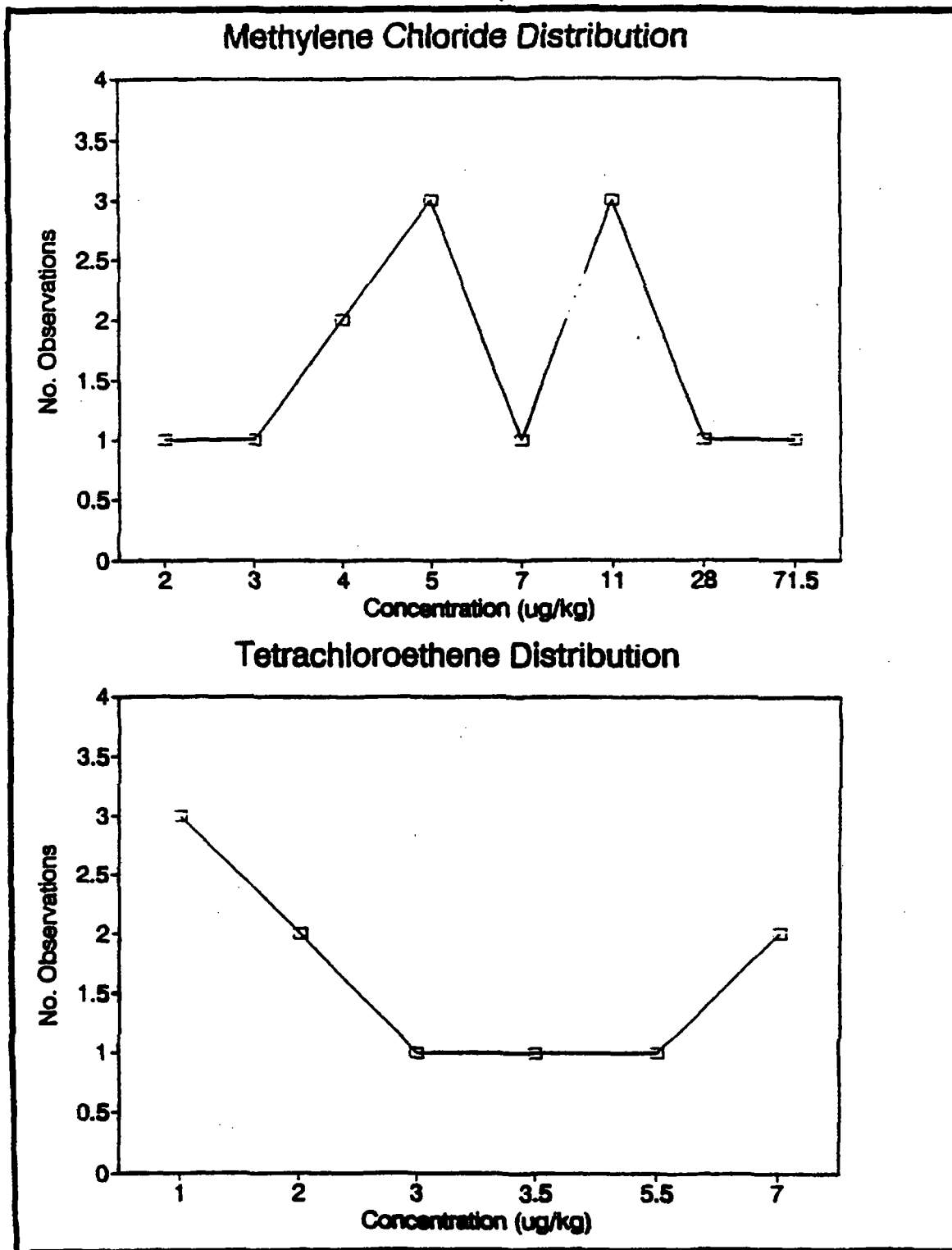
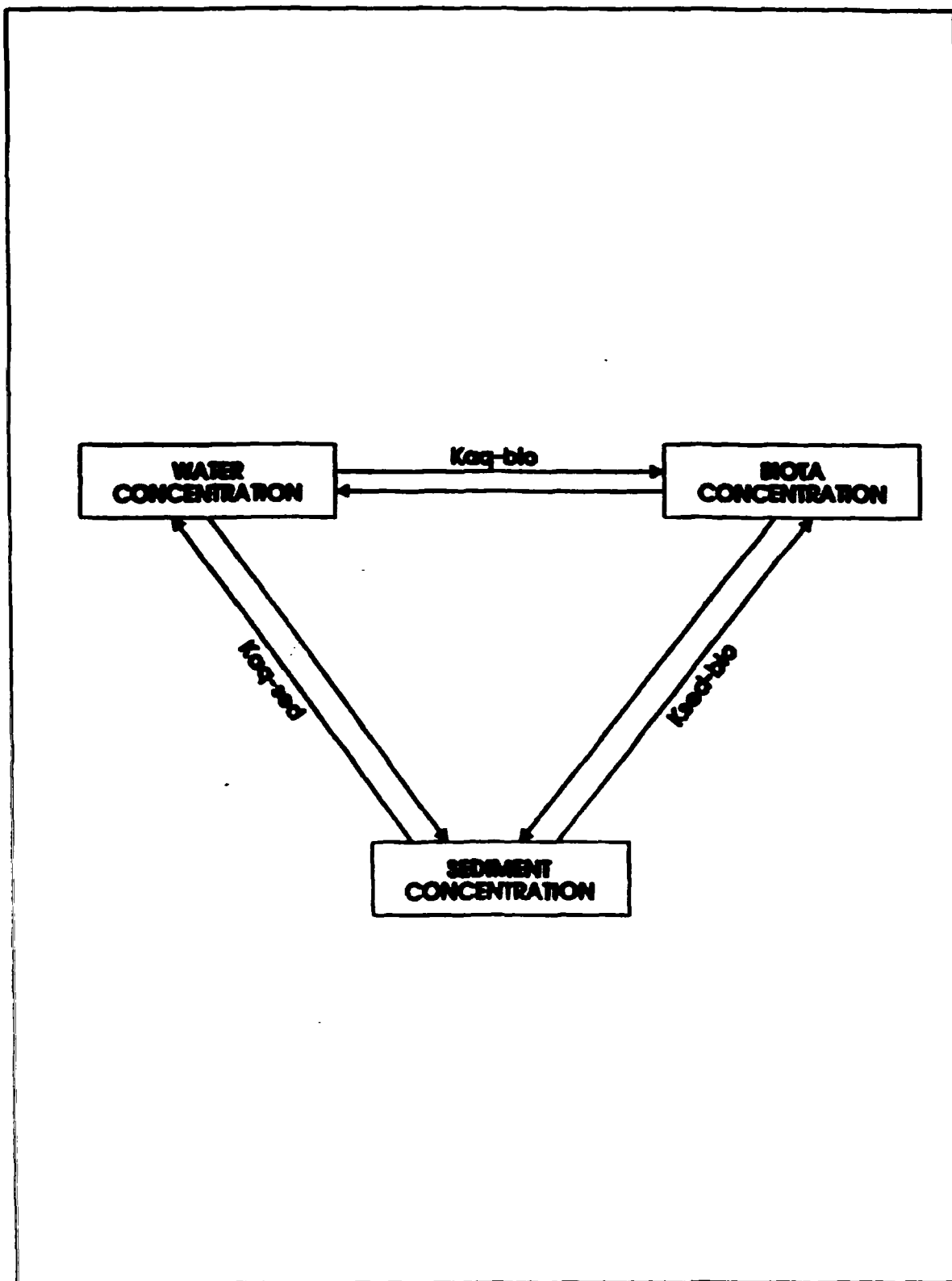


FIGURE 6-2 DISTRIBUTION OF TWO COPCs

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FIGURE 6-3
EQUILIBRIUM PARTITIONING

7. SUMMARY AND CONCLUSIONS

This section presents a summary and conclusions of the findings of the RI for the Conrail Site. The investigation was completed in three phases. The interpretations of the findings for the hydrogeology, nature and extent of contamination, fate and transport, and risk assessment are also summarized. The RI was performed by E & E under contract to EPA Region V. The information obtained will be used to evaluate a range of feasible, permanent solutions for remediation as part of the FS.

7.1. SITE PHYSICAL HYDROGEOLOGY

The aquifer of concern is a glacial outwash deposit consisting of sand, and sand and gravel. The glacial outwash material is approximately 150 feet thick and lies on shale bedrock that is not a water-bearing unit. Depth to water varies between 3 and 20 feet BGS in the study area. The general flow direction throughout the depth of the aquifer is to the west-northwest, except in the LaRue Street residential area, where the general flow direction is north. Hydraulic conductivity values derived from slug test data were used in addition to measured horizontal gradients to estimate the horizontal flow velocity of groundwater. Heterogeneity in site conditions caused variability in input parameters that result in a velocity range for groundwater of 11 feet per year to 2,200 feet per year. The mean horizontal flow velocity of groundwater is 200 feet per year.

7.2 SOIL SAMPLING RESULTS

A subsurface soil investigation was conducted as part of the field investigations in order to determine the nature and extent of identified and suspected source areas contributing

to identified groundwater contamination. Based on analytical results from subsurface soil samples, two source areas on the Conrail facility have been identified.

A CCl_4 source area was identified in the eastern section of the classification yard near track 69 in the saturated zone. The dimensions of this source area are approximately 75 feet by 30 feet by 7.5 feet. The analytical data from these subsurface soil samples suggests that this CCl_4 source area extends beyond the boundaries established to date. CCl_4 contamination was also detected ($16 \mu\text{g}/\text{kg}$) in a 128 to 130-foot sample interval, but pooled residual CCl_4 was not detected on top of bedrock.

A TCE source area was identified in the western section of the classification yard between tracks 65 and 66. The approximate dimensions of this source area are 120 feet by 10 feet by 4 feet. This source area is in the unsaturated zone.

CCl_4 contamination was detected in the saturated zone in subsurface soil samples collected from the receiving yard on the Conrail facility, which is upgradient of the LaRue Street groundwater contamination area. CCl_4 concentrations were less than or equal to $31 \mu\text{g}/\text{kg}$. This contamination is thought to contribute to detected CCl_4 groundwater contamination downgradient of this area.

7.3 GROUNDWATER MONITORING WELL SAMPLING RESULTS

CCl_4 and TCE contamination were detected in groundwater samples collected from monitoring wells screened in the shallow, intermediate, and deep zones on the Conrail facility, at the St. Joseph River, and in the area between these points, in the County Road 1 plume. Groundwater flow direction in all three zones is west-northwest, and the groundwater contamination plume follows this path from the Conrail facility to the St. Joseph River. Groundwater samples collected from monitoring wells located hydraulically upgradient of the plume and the site in the shallow and intermediate zones did not detect any VOC contamination. LSA samples collected hydraulically upgradient in the deep zone did not detect any VOC contamination.

The maximum concentration of CCl_4 groundwater contamination was $110,000 \mu\text{g}/\text{L}$ collected from monitoring well MW46S located in the track 69 source area. This concentration is 13.8% of the solubility of CCl_4 and suggests a CCl_4 DNAPL source (EPA

1992a). Site background information and the detection of CCl_4 in a subsurface soil sample at 130 feet BGS also suggest that the CCl_4 source area in track 69 is a DNAPL source.

The maximum concentration of TCE detected in a groundwater sample was from monitoring well MW41, located immediately downgradient of the Conrail facility. This concentration is 1.4% of the solubility of TCE and suggests a TCE DNAPL source. MW41 is side-gradient of the TCE source area identified in the classification yard. Based on analytical and hydrologic data, the DNAPL source is on the Conrail property, but is, as yet, unidentified.

CCl_4 was detected only in groundwater samples collected from shallow monitoring wells in the LaRue Street plume surrounding the identified CCl_4 soil contamination in the receiving yard on the Conrail facility. The soil contamination is thought to contribute to the identified CCl_4 groundwater contamination. Monitoring wells upgradient of the soil contamination did not detect CCl_4 .

TCE was detected only once (MW20S - 15 $\mu\text{g/L}$) above the MCL for TCE in the LaRue Street plume. TCE was detected below the MCL (5 $\mu\text{g/L}$) throughout the plume. Monitoring wells located upgradient of the Conrail facility did not detect TCE. Based on groundwater analytical data, the source of the TCE contamination is on the Conrail property, but is, as yet, unidentified.

7.4 DRAINAGE NETWORK SAMPLING RESULTS

Analysis of the sediment and water samples from the Conrail drainage network revealed no detectable levels of TCE or CCl_4 . The data suggest that the drainage network is not currently a source of the identified groundwater contamination, which resulted from surface spills on the facility. The possibility that the drainage network historically acted as a conduit for VOC contamination cannot be assessed due to lack of previous sampling data from the network.

7.5 FATE AND TRANSPORT

A focused discussion on the subsurface and groundwater fate and transport of CCl_4 and TCE is presented. Processes such as volatilization, liquid transport, sorption, and transformation reactions have likely occurred at the site based upon the contaminants present

and the observed environmental conditions. Volatilization of the dissolved chlorinated aliphatic hydrocarbons from the groundwater that is present at or near the water table can transfer significant contaminant mass from groundwater to soil gas. There is a high probability that this process has occurred and is currently operative in the study area. Liquid transport is occurring at the site as dissolved contaminants in groundwater undergo migration. If DNAPL is present, it may also migrate by density driven-liquid transport processes specific to DNAPLs. Sorption of chemicals of potential concern onto aquifer materials is expected to be an important process at the site. Analytical results of total organic carbon in the aquifer material are combined with chemical-specific data to quantify sorption so that retardation can be estimated. The retardation is used to estimate the migration rate of contamination relative to the groundwater flow velocity. Of the numerous transformation reactions that may possibly occur, sequential reductive dehalogenation is important because it appears to be functioning at the site.

The fate and transport processes coupled with site-specific data enable the estimation of movement, mass, and loading of CCl_4 and TCE. The heterogeneity in site conditions causes a large range in the time (6 to 1,200 years) estimated for groundwater to travel from the source areas to the St. Joseph River. If retardation does not take place, contamination would undergo advection at the same rate as groundwater flow. If available sorptive capacity in the aquifer materials permits sorption, it is estimated that CCl_4 and TCE will travel at approximately 40 percent of the rate of groundwater. The total mass of CCl_4 and TCE remaining in the aquifer as dissolved contaminants in the groundwater and sorbed to aquifer materials can be estimated based on analytical data from the site and an estimate of DNAPL volume. The estimated total mass of CCl_4 and TCE in the groundwater and sorbed to aquifer materials is 20,000 pounds. Residual DNAPL may contribute 150,000 pounds as CCl_4 and TCE, combined. The estimated loading from the site to the St. Joseph River is 20 pounds of TCE and 20 pounds of CCl_4 per year. Assuming this loading, it would take 200 years to remove the CCl_4 and TCE that is dissolved in the groundwater presently in the aquifer. If DNAPL is present, this estimate of elapsed time for natural attenuation would be much greater.

Transformation reactions resulting in the formation of daughter products of CCl_4 and TCE are occurring at the site. The daughter compounds chloroform and 1,2-dichloroethene

(total) were detected in monitoring well samples that also contained higher concentrations of CCl_4 and TCE, respectively.

7.6 HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENT

Based on the information, observations, and analytical data gathered during the three phases of the RI/FS, the baseline risk to human health and the surrounding ecological environment was assessed and is summarized below.

7.6.1 Assessment of Human Health Risk

The human health assessment examined the risks to railyard workers and residents of nearby neighborhoods from exposures to Conrail Site contaminants. The 19 chemicals selected as contaminants of potential concern were all volatile organic compounds.

Potential risks to railyard workers were evaluated for existing conditions and for a possible future excavation scenario. Only the vapor inhalation pathway was evaluated under the current risk scenario. The potential excess cancer risk for the RME case for this pathway is 3.40×10^{-3} , which exceeds EPA's threshold value range (10^{-6} to 10^{-4}) for upper-bound lifetime cancer risks to an individual. The hazard index for the RME case for this pathway is 253, which indicates that there is potential for adverse noncarcinogenic risks via this pathway.

The potential future risks to railyard workers were evaluated via the soil contact pathway (direct contact and incidental ingestion) and inhalation of airborne contaminants. These pathways were evaluated for a possible excavation scenario, in which it is envisioned that current subsurface contamination is exposed and then becomes surface contamination. The potential excess cancer risk and the potential adverse noncarcinogenic risk calculated for the soil contact pathway exceed their associated EPA threshold values. This indicates that there is a potential for risks to workers during excavation activities.

Two distinct plumes of groundwater contamination, designated as Plume 1 (County Road 1 Plume) and Plume 2 (LaRue Street Plume), respectively, are migrating from the railyard towards the St. Joseph River. Plume 1 is migrating north and west of the facility, while Plume 2 is migrating north and east of the facility. Plume 1 contained 10 COPCs while Plume 2 contained seven COPCs. The potential risks posed by each plume varied based not

only on contaminant concentration, but also based on the toxicological properties of the chemicals detected in the plumes, and on the pathway of exposure.

Residential exposures from groundwater usage and inhalation of indoor airborne contaminants from groundwater vapors was evaluated for both plumes. The residential groundwater usage pathway included ingestion of drinking water, dermal contact with water during showers, and inhalation of water vapors during showers. The inhalation of indoor airborne contaminants from groundwater vapors were evaluated based on an empirical model of groundwater contaminants volatilizing into nearby unsaturated soil gas, infiltrating into basements and subsequently diffusing throughout the entire home.

The potential risks from these exposures were greater in Plume 1. The excess cancer risk for the RME case of groundwater usage is 9.00×10^{-3} . The excess cancer risk associated with inhalation of indoor airborne contaminants for the RME case was 1.71×10^{-4} . These values exceed the EPA threshold value range of 10^{-6} to 10^{-4} . The potential for noncarcinogenic adverse risks from residential exposures to contaminants in Plume 1 was above the EPA hazard index threshold level of 1 for the groundwater usage pathway (hazard index = 201). The potential cancer risks in Plume 1 are due to exposure to carbon tetrachloride, chloroform, 1,1-DCE, trichloroethene, and vinyl chloride. The potential adverse noncarcinogenic risks due to Plume 1 groundwater usage is due solely to carbon tetrachloride.

Residential exposures to Plume 2 contaminants results in potential excess cancer risks from both groundwater usage and inhalation of airborne contaminants from groundwater vapors. The RME case risk via the groundwater usage pathway is 1.45×10^{-3} . The RME case risk via the indoor air inhalation pathway is 1.11×10^{-5} .

The hazard index for the Plume 2 groundwater usage pathway is 4.16. This indicates that there is a potential for long-term, chronic adverse noncarcinogenic risks to individuals.

The excess cancer risks for Plume 2 groundwater usage and indoor air inhalation pathways are due to benzene, carbon tetrachloride, chloroform, and trichloroethene. The noncarcinogenic risks in Plume 2 are due to carbon tetrachloride and acetone.

7.6.2 Ecological Assessment

The ecological assessment examined existing and potential risks posed by site-derived contaminants to nearby natural habitats and associated flora and fauna. The primary focus of the risk assessment was on aquatic organisms in Baugo Bay, the St. Joseph River, and the Conrail ponds.

Four special species of concern were identified within 2 miles of the Conrail Site. Two were plant species whose latest sightings were in the 1930s and the 1940s. One mammal (*Taxidea taxus*) has been sighted recently north of the St. Joseph River, which is an area that is probably not affected by the site. The Cooper's Hawk (*Accipiter cooperii*) has been recently sighted near the site. However, the hawk prefers to nest in mature trees, will not hunt within 0.5 miles of the nest, and will travel up to 2 miles to hunt over open fields and along woodland margins. Therefore, it is unlikely that site-related contaminants will effect the Cooper's Hawk.

Surface water CPECs were chosen based on comparisons of surface water sample analytical results with current ambient water quality criteria. Lead was chosen as a surface water CPEC in the St. Joseph River, while lead and zinc are CPECs for the ponds. No surface water CPECs were selected for Baugo Bay.

Organic CPECs in sediments were chosen based on comparisons of sediment sample analytical results with sediment quality criteria. For those chemicals that did not have published values, the criteria were calculated based on the equilibrium-partitioning approach for non-polar organics. No organic CPECs were chosen for Baugo Bay or for the St. Joseph River. The five CPECs chosen for the ponds are 4,4'-DDT, 4,4'-DDD, 4-methylphenol, fluorene, and Aroclor 1254.

Many inorganic contaminants were detected in Baugo Bay, the St. Joseph River, and in the ponds. A statistical comparison of the inorganic results from the background locations and the results from the downstream locations for Baugo Bay and the St. Joseph River indicates that there was no significant difference between the concentrations of contaminants in the two locations. Therefore, the contamination does not appear to be site-related. There were no background ponds available for similar statistical comparisons for the pond data. Therefore, CPECs chosen for the pond area are arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, and zinc.

Sediment hazard indices (the ratio of the sample concentration to the toxicity reference value) were greater than 1 for cadmium, copper, lead, zinc, fluorene, and 4-methylphenol. Arsenic, chromium, manganese, mercury, nickel, DDT, DDD, and Aroclor 1254 also had indices greater than 1, but the magnitudes of the indices were lower than for the previously stated CPECs. The maximum concentrations of these CPECs were all located in the westernmost pond.

The surface water hazard index for lead is greater than 1 while the hazard index for zinc is less than 1. As in the sediment CPECs discussed above, the highest concentrations of surface water CPECs were found in the westernmost pond.

Potential risks to benthic organisms from CPECs present in the ponds are assessed to be moderate to high. CPECs were detected only in the westernmost pond, which receives drainage from an outfall of the facility drainage system. This pond is less than 1 acre, and other suitable habitats are available for use in the area. Non-benthic organisms that use the pond for a portion of their life cycle may be potentially impacted by the sediments, but they are at a lower risk than benthic organisms. The terrestrial ecosystem in the general vicinity of the site does not appear to be impacted by site-related CPECs.

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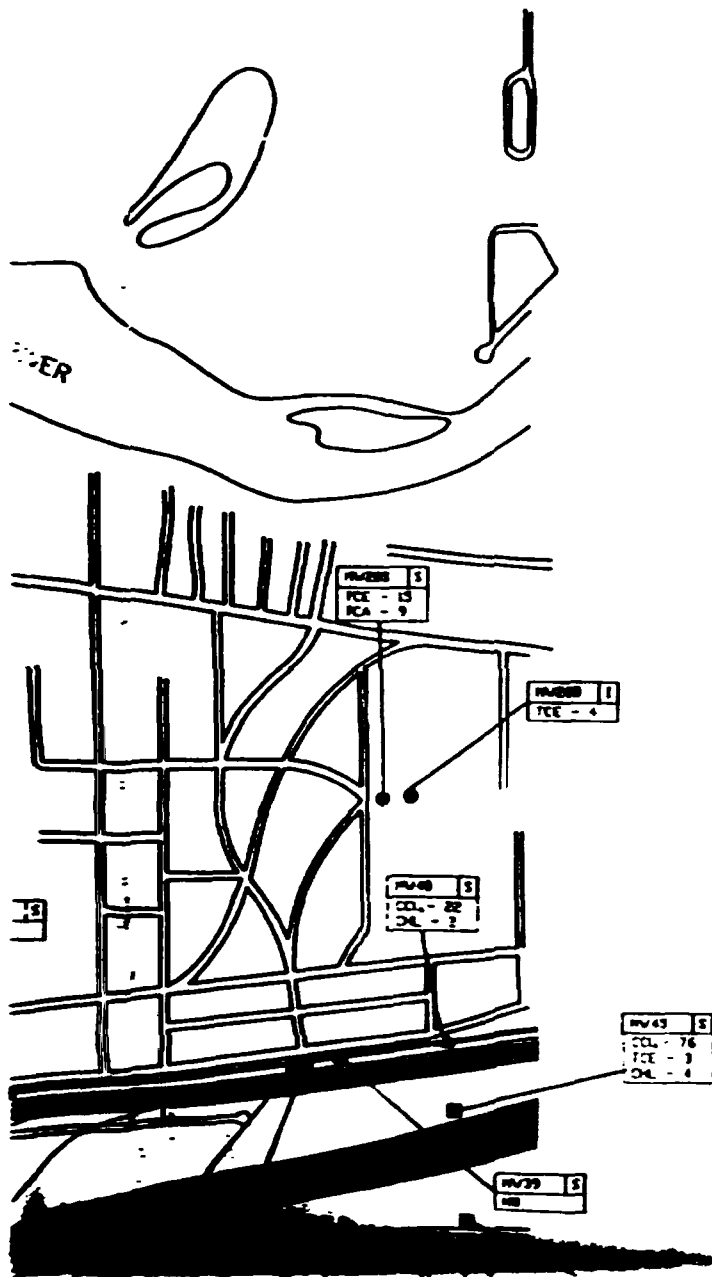
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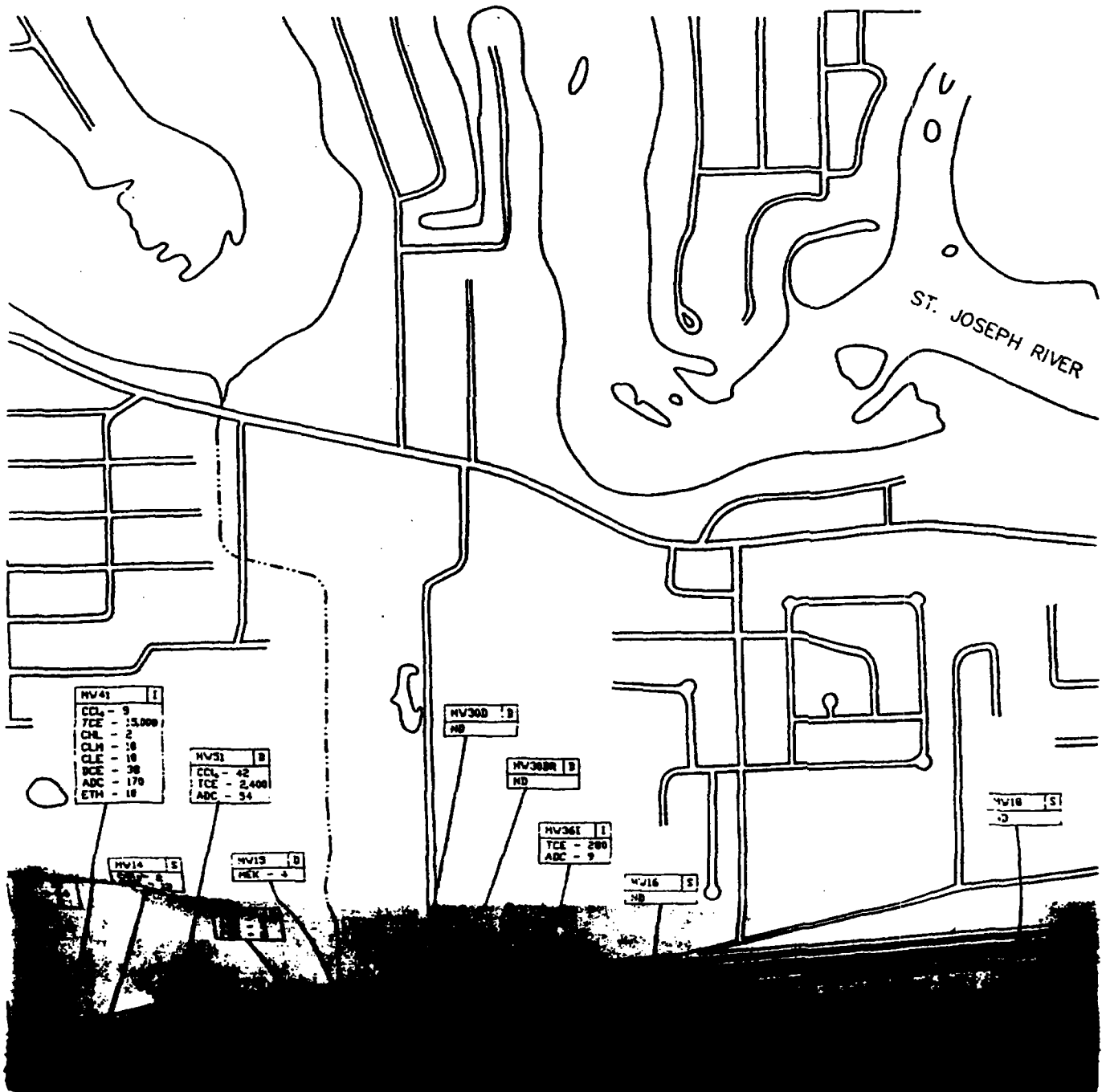
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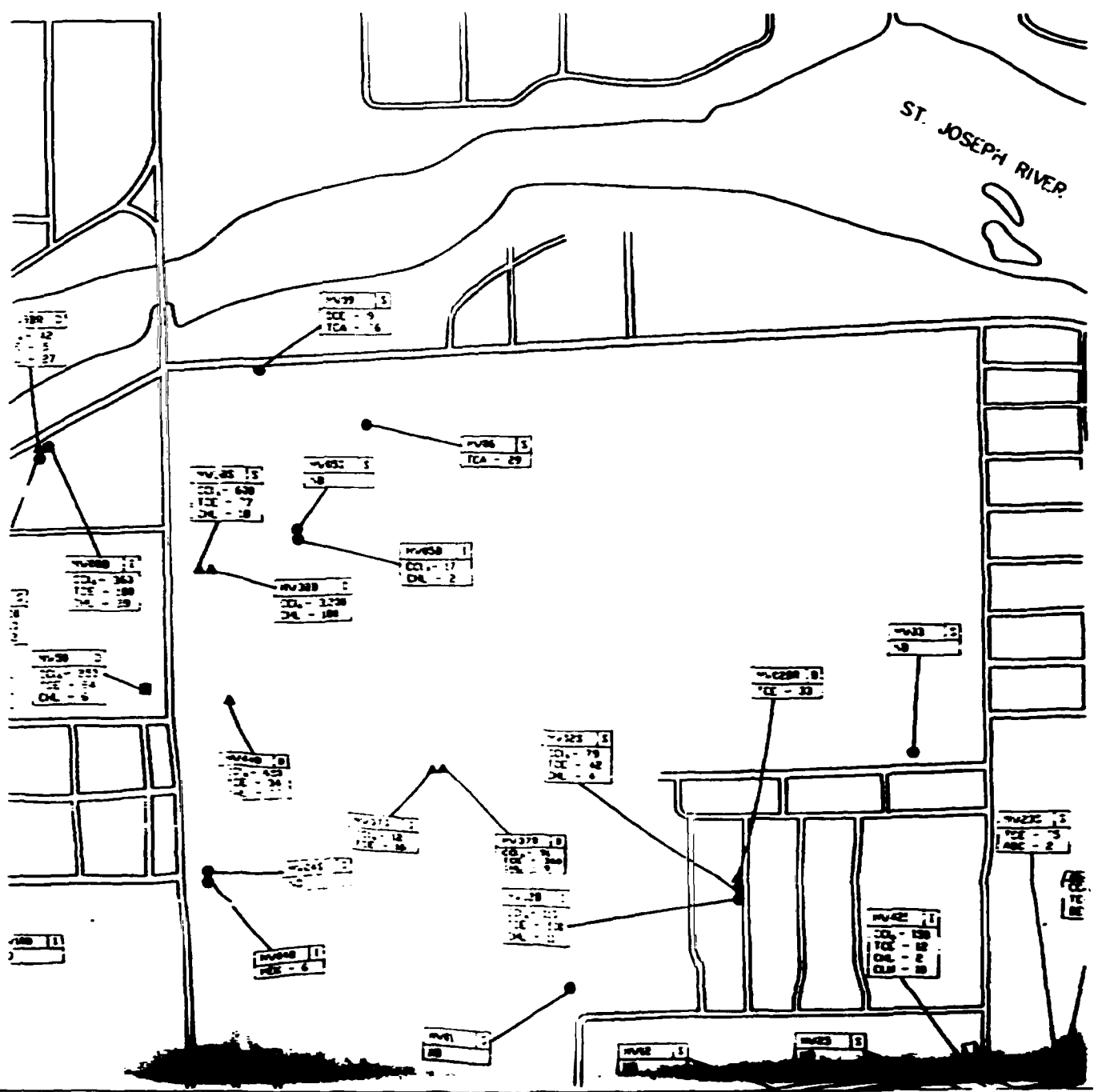
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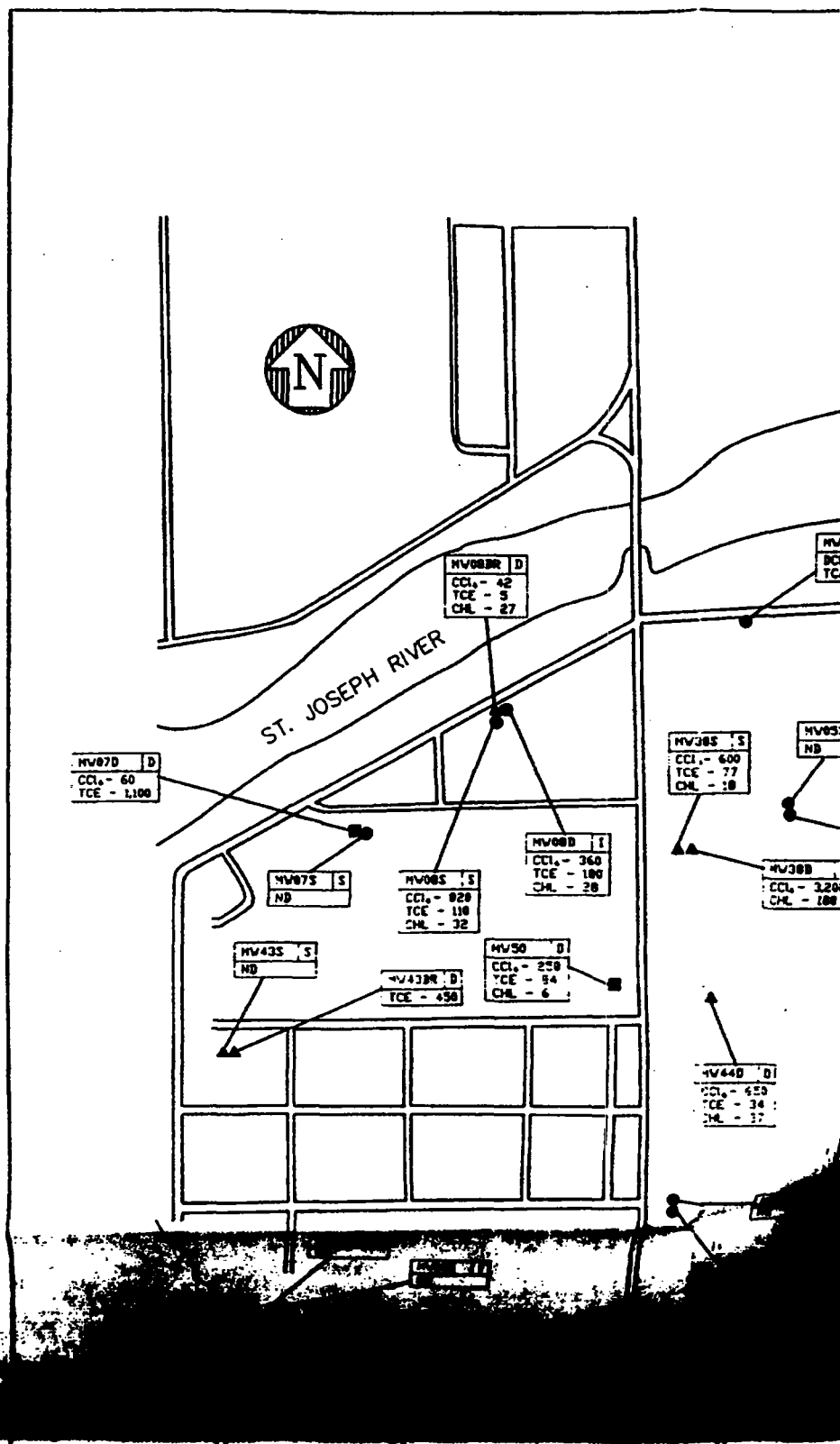
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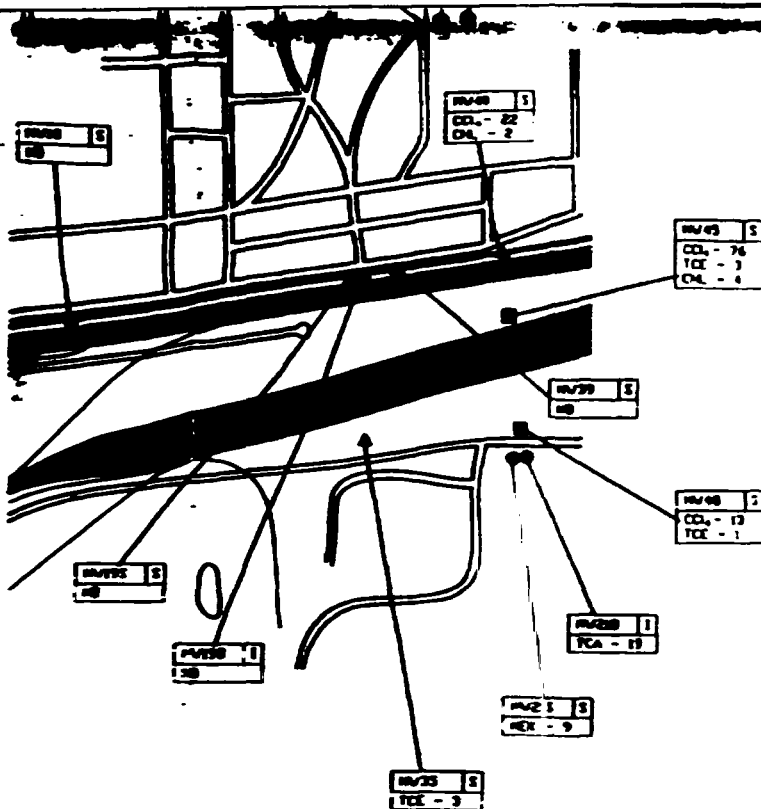






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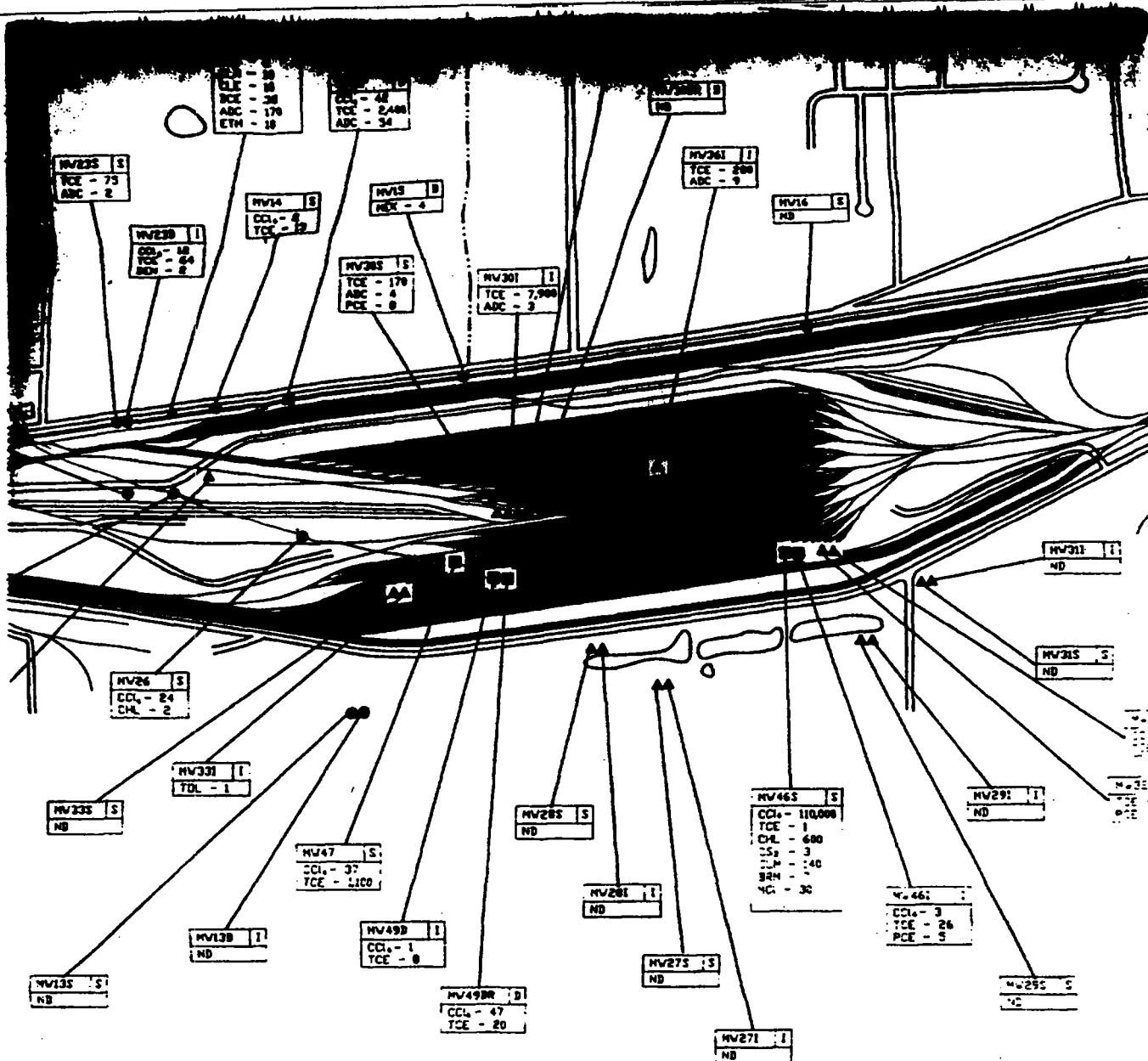





**VOLATILE ORGANIC COMPOUND
RESULTS FOR PHASE III GROUNDWATER
SAMPLES (CONCENTRATIONS IN ug/L)**

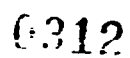
CONRAIL SITE

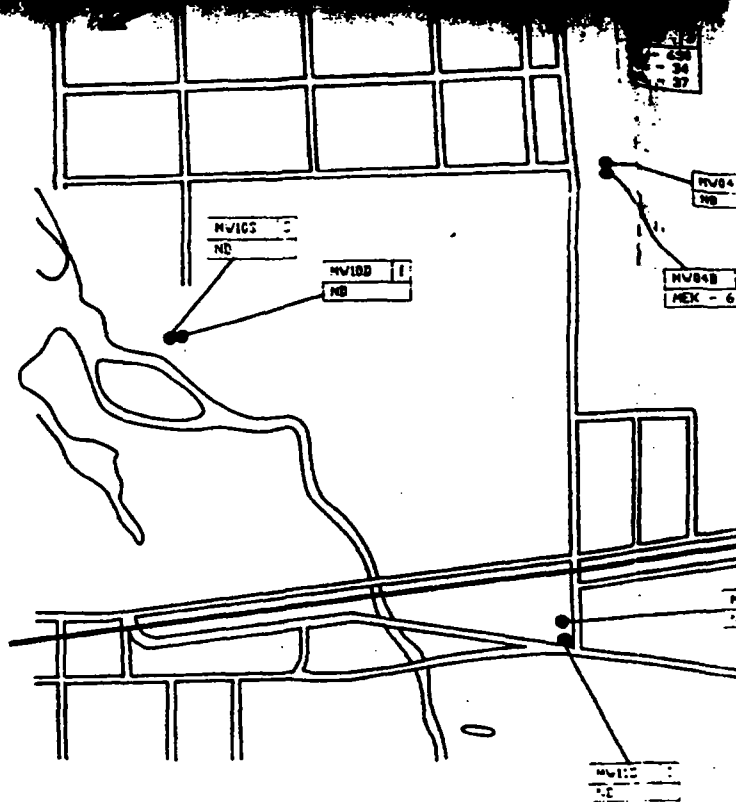
SCALE	DATE ISSUED	C.A.B. FILE NO.	ISSUING NO.	REV.
1" = 600'	MAY 1993	273RPLT1	PLATE 1	0



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DESIGNED BY:	CHECKED BY:
B. LOMBARDI	R. HACKLER
DRAWN BY:	APPROVED BY:
B. KROULDIS	B. LOMBARDI

DATE	OWN	APP'D	DESCRIPTION
			REVISIONS





MONITORING WELL
NUMBER

MONITORING ZONE
S - SHALLOW
I - INTERMEDIATE
D - DEEP

COMPOUND

MW04B I
MEK - 6

CONCENTRATION

PLOTTED: MAY 1993